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ASSESSMENT

OF THE

EFFECTIVENESS AND EFFECTS

OF

LAND DISPOSAL METHODOLOGIES

OF

WASTEWATER MANAGEMENT

DEPARTMENT OF THE ARMY
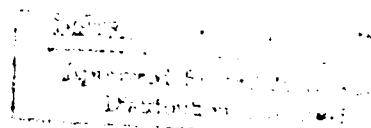CORPS OF ENGINEERS

JANUARY 1972   WASTEWATER MANAGEMENT REPORT 72-1

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PREFACE

Pilot studies for comprehensive regional wastewater management
were begun by the Corps of Engineers in early 1971. These studies
were an outgrowth of comprehensive water resource planning efforts
the Corps already had underway and were set up in two stages. The
first was a broad feasibility phase; the second, a more detailed
survey phase.

To note progress on the pilot studies, the Corps recently completed
the first, or feasibility,phase covering five metropolitan regions;
the Merrimack River Basin north of Boston in New England, the
Cleveland, Detroit, and Chicago metropolitan areas on the Great Lakes,
and the San Francisco Bay area in California. The pilot studies are
being carried out with the cooperation and participation of the
affected Federal, State and local planning agencies.

Following the feasibility stage it was determined that the subject
of land treatment of wastewater needed considerable attention to
refine specific considerations of treatment effectiveness, optimal
sites, engineering design criteria and environmental impacts as well
as the general degree of confidence the Corps officials should have
in such techniques. Further, the planners and design engineers
required greater insights into the mechanisms that provide the bases
for land treatment before they could properly manipulate and apply
the methodology. The emphasis on plan evaluation had highlighted
weaknesses in the ability to produce technically adequate plans for
land treatment systems. Such determinations were derived partially
from contract evaluations included in the feasibility reports and
partly from in-house assessments. The reasoning behind the stated
needs included concern over the implications of system failure,
problems of public acceptance and the unfamiliarity of the study
personnel with such techniques.

The following report represents the efforts of an independent,
interdisciplinary group of consultants at the University of Washington,
Seattle, Washington to provide such an assessment. The report should
therefore not necessarily be construed to represent the views of the
Corps of Engineers.

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20. ABSTRACT *(Continue on reverse side if necessary and identify by block number)*

The principal goal of this study was to prepare a comprehensive assessment of the fiasibility and effects of land disposal of secondary treated waste water composed of municipal and industrial waste discharges. The assessment is provided herewith in the form of a report which is designed as an educational document and is intended to be useful to planning units of the U.S. Army Corps of Engineers. This report attempts to identify the scientific basis for treatment effectiveness and to offer explicit professional judge-

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ments regarding the extent of, treatment effectiveness by land disposal.

ASSESSMENT OF THE EFFECTIVENESS AND EFFECTS OF
LAND DISPOSAL METHODOLOGIES OF WASTE WATER MANAGEMENT.

by

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Jan————, 1972

TABLE OF CONTENTS

Table of Contents (Continued)   <u>Page</u>

CHAPTER I - PURPOSE AND SCOPE

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The principal goal of this study was to prepare a comprehensive assess-
ment of the feasibility and effects of land disposal of secondary treated
waste water composed of municipal and industrial waste discharges. The
assessment is provided herewith in the form of a report which is designed
as an educational document, and is intended to be useful to planning
units of the U. S. Army Corps of Engineers.

The study represents a state-of-the-art evaluation of the environmental
impacts associated with secondary treated waste water disposal by spray
irrigation, rapid infiltration ponds and overland runoff. Special empha-
sis was given to the prediction of probable environmental impacts for
ecosystems typical of study areas previously considered by the Corps of
Engineers, (San Francisco Bay, Chicago-SouthEnd Lake Michigan, South-
eastern Michigan, Cleveland-Akron Metropolitan and the Merrimack River
Basin areas). In addition to providing a narrative discussion of overall
ecosystem response to the addition of waste water by typical land dis-
posal techniques, the report attempts to identify the scientific bases
for treatment effectiveness and to offer explicit professional judgements
regarding the extent of treatment effectiveness by land disposal. Specific
products of the study requested by the contractor are described in
Appendix A. The scope of the treatment offered herein was expanded by
the study group beyond the scientific and mechanistic intents covered in
Appendix A to include questions of overall engineering feasibility of
land disposal methodology. These evaluations were provided by two
additional consultants (Drs. Loehr and Mar) and their comments are in-
cluded in Appendix B.

CHAPTER II - METHOD OF APPROACH

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A. Conference and Seminar Procedures

An interdisciplinary team (Table II-1) was assembled by the contract
officer from the faculties of the Colleges of Forest Resources and
Engineering at the University of Washington. It should be noted that
the period of time available to this team, from the formation of the
study team to the deadline for submission of the input, was less than
eight weeks. The study team found that commitment to intensive, regular
seminars was a useful working style in light of the severe time con-
straint placed on the project. Initially, the seminars were devoted to
problem and role definition by the participants and later were used as
a forum for discussion with outside consultants (Table II-2). Close
liaison with the Corps of Engineers was maintained via the active parti-
cipation of two members of the Seattle District Office (Urabeck and
Hogan) in the seminar sessions. All pertinent background reports were
made available by the Corps of Engineers and they were combined with
additional research literature into a working library for the study team
by Dr. Driver (Appendix C).

B. Baseline Decisions

Early in its deliberation, the study team found it necessary to assume a
baseline chemical and biological quality for secondary treatment plant
effluent. The average quality agreed upon (Table II-3) was based upon
published data and the professional judgement of the team members. In
general, it was assumed that the waste water would be generated by both
municipal and industrial sources subjected to standard secondary treat-
ment processes and chlorinated to such a degree that the final effluent
would not be toxic to plant life. The hazards in attempting to assume
average qualities for mixed municipal and industrial effluents are ob-
vious, and the reader is cautioned that judgements offered in later

Table II-I

PROJECT PARTICIPANTS

Dr. Charles H. Driver, Professor (Forest Resources), Soil Biology

Dr. Bjorn F. Hrutfiord, Associate Professor (Forest Resources), Organic Chemistry

Dr. Demetrios Spyridakis, Research Assistant Professor (Engineering), Water Chemistry

Dr. Eugene B. Welch, Associate Professor (Engineering), Aquatic Biology

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Mr. Dwain Hogan, Chief, Hydraulics Section, Seattle District, U.S. Army Corps of Engineers

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Ms. Elaine Ramel, Project Secretary

Capt. Steven F. Dice, Project Supervisor, U.S. Army Corps of Engineers, Washington, D.C.

Dr. Russell F. Christman, Associate Professor (Engineering), Project Coordinator

Table II-2

ADDITIONAL PROJECT PARTICIPANTS

Dr. Brian W. Mar, Research Professor (Engineering), University of Washington, Engineering Feasibility

Dr. Raymond C. Loehr, Professor (Engineering), Cornell University, Engineering Feasibility

Dr. William E. Sopper, Professor (Forestry), Pennsylvania State University, Land Disposal Practice and Effects

Dr. Nels R. Benson, Chemist, Tree Fruit Research Center, Wenatchee, Washington, Retention of Trace Metals in Soils

Dr. Paul Hughes, Ground Water Hydrologist, USACE, Sacramento, California, Hydrologic Aspects of Land Disposal

Mr. Sherwood Reed, Sanitary Engineer, U.S. Army Cold Regions Research and Engineering Laboratory

Mr. Richard E. Thomas, Research Soil Scientist, Robert S. Kerr Water Research Center (EPA), Ada, Oklahoma

Table 11-3

ASSUMED QUALITY OF SECONDARY EFFLUENT

Material	Concentration mg/l	Quantity* lbs/ac/yr
Biochemical Oxygen Demand (BOD)	25	589
Chemical Oxygen Demand (COD)	70	1356
Suspended Solids	25	589
Microorganisms; typical of municipal secondary effluent:		
pH	7.0 ± 0.5	--
Nitrogen (Total)	20 as N	470
Phosphorous (Total)	10 as P	235
Trace Metals:		
Chromium	0.02 to .14	0.5 to 3.0
Copper	0.07 to .14	2.0 to 3.0
Iron	0.10 to 4.3	2.4 to 100
Lead	0.01 to .03	0.2 to .72
Manganese	0.20	4.8
Nickel	0.03 to .20	0.7 to 4.8
Zinc	0.20 to .44	4.8 to 5.0
Cadmium	0.015	0.03
Mercury	0.01	0.24
Boron	1.0	24
Chloride	45	1058
Sodium	50	1175
Calcium	24	565
Magnesium	17	398
Potassium	14	328

*Assuming 2 inch/week application rate 52 weeks per year

sections of this report are, unless otherwise stated, based upon the assumed chemical and biological qualities of secondary effluent shown in Table II-3. In the case of organic chemical composition it was useful to construct a list of compounds which, in the professional opinions of the team members, would serve as a model composition to be expected from diverse municipal and industrial sources.

Baseline values on the nature and concentration of organic compounds in mixed domestic and industrial secondary treated sewage effluent are not available in the literature. It was decided to develop a list of classes of organic compounds from the scanty literature data, then select a model for each class for consideration in the land disposal effectiveness evaluation. Concentrations were to be established and the sum of these would correlate with the baseline BOD and COD values. The first two objectives have been met, however, it is difficult to establish meaningful concentration values due to lack of information. The baseline classes, models, and some concentration information are presented in Table II-4. A complete list of organics reported in waste water is included in the Appendix D.

This report covers three alternatives in waste water disposal systems for secondary effluent: Spray irrigation, rapid infiltration ponds, and overland runoff. Each system has particular requirements for hydraulic conductivity of soils and chemical properties. Maximum chemical renovation of waste water is desired with minimum deleterious effects on the soils of the disposal site. Most conditions suggest a continuous cover of vegetation necessary to prevent soil erosion and clogging of the soil surface.

Spray Irrigation: The design system for spray irrigation suggests an application rate of up to 2" of waste water per week continuously during the season when air temperatures are not significantly below freezing. The first requirement for a successful spray irrigation system is soils with infiltration and percolation capacities sufficient to handle the design loading of 2" of waste water in an eight hour period (0.25 inch/hour).

ASSUMED QUALITY OF SECONDARY EFFLUENT FOR
BASELINE ORGANIC COMPOUNDS

Class	Model	Concentration mg/l	Quantity* lbs/ac/yr
Humic substances	Naturally colored water	40	940
Carbohydrates	Glucose	1	24
Proteins/Amino Acids	Creatinine	4	96
Detergents	ABS/LAS	10	235
Organic Acids	Butyric Acid	3	72
Purines	Uric Acid	3	72
Neutral Hydrocarbons	Kerosene	2	48
Aromatic Hydrocarbons	Toluene	1	24
Polynuclear Hydrocarbons	Naphthalene	0.1	2.4
Plasticizers	Di-2-ethylhexylphthalate	1	24
Nitrogen Compounds	Pyridine	0.1	2.4
Chlorinated Hydrocarbons	Hexachlorophene/DDT/PCB	0.001	0.002
Polyphenolics	Lignin/Tannin	1	24
Resin Acids	Dehydroabietic	trace	trace
Phenolics	Phenol	1	24
Terpenes	Limonene	trace	trace

*Assuming 2 inch/week application rate 52 weeks per year

Soils of a silt loam texture with reasonable aggregation such as the
Herman-Hinckley series in New England, the Miami series in the Lake
States and the Yolo series in the Bay area, all have sufficient hydraulic
capacity to handle waste water at the design loading. Ideally, the soils
should be uniformly about 5' deep to either bedrock or the water table.

In the more humid areas of the Lake States and New England spray irriga-
tion could continue to minimum air temperatures of approximately 25° as
long as surface soil temperatures are not below 32°. A continuous mat of
vegetation will provide insulation for the soil and continuous melt even
if thin ice layers occur. In dryer regions with lower dew point tempera-
tures rapid evaporation losses might super cool water so that spray irri-
gation might be discontinued when air temperatures approach 32°.

The slope of land for spray irrigation is not critical as long as infiltra-
tion rates are sufficient for all water to enter the soil profile.

Rapid Infiltration Ponds: The rapid infiltration pond has a suggested de-
sign loading of up to 330'/year. Water is applied to the ponds for 10-14
days, during which time anaerobic conditions exist. The pond is then
allowed to dry for a short drying cycle for oxidation of organic matter
and maintenance of rapid infiltration. Grasses have been found to be
beneficial to prevent clogging of soil pores even for rapid infiltration.
Loading rates are approximately 1'/day for the days of operation of the
pond. Sustained percolation rates of 1/2"/hour are necessary to achieve
the desired loading of waste water. A much coarser soil in the sandy to
sandy gravel texture would be required. Such a soil would have a tremen-
dously reduced active surface area and require several hundred feet of
contact to achieve the same degree of renovation that 5' of spray irrigation
column might. This distance may either be vertical or lateral depending on
prevailing soil conditions.

Overland Runoff: The overland runoff system may be applied only on imper-
vious soils (.2"/day). Soils of this nature are usually of very heavy texture

- 8 -

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in both the surface and subsurface. Clay soils have a very high reactive
surface area but extremely low hydraulic conductivity rates. Renovation
of the waste water is achieved by movement of water over the soil surface
and through the decaying material. A continuous vegetative cover is very
necessary to achieve the desired degree of filtration and physical removal
of BOD and suspended solids. The overland runoff system requires trenches
spaced approximately 100' apart per percent of slope to achieve a desired
retention and contact time with the soil. Slopes may vary between 2-6%.
Spray water is applied intermittently with frequent 1-5 day drying periods.

Assumptions regarding land areas and water quantities for each of the dis-
posal alternatives are described in Tables 11-5 and 11-6.

Table II-5

DAILY WASTE WATER PRODUCTION OF CITIES OF
VARYING POPULATIONS
(assuming 100 gal. per person per day)

Population (thousands of persons)	Waste Water Production Per Day		
	cubic feet (thousands)	acre feet	gallons (millions)
1	13.4	0.31	0.1
5	66.8	1.53	0.5
25	334.2	7.67	2.5
50	668.4	15.35	5.0
75	1,002.6	23.02	7.5
100	1,336.9	30.69	10.0
500	6,684.5	153.45	50.0
1,000	13,369.0	306.91	100.0

Table 11-6

LAND REQUIREMENTS FOR VARYING POPULATIONS
(assuming 100 gal. per day/person of waste water production)

Population	Spray 1/ Irrigation		Infiltration 2/ Pond		Overland 3/ Runoff	
(thousands of persons)	Acres	Sq.Mi.	Acres	Sq.Mi.	Acres	Sq.Mi.
1	12.9	0.02	0.4	--	14.	--
5	64.5	0.1	2.0	--	69.	0.1
25	322.	0.5	10.2	--	347.	0.5
50	644.	1.0	20.4	--	694.	1.1
75	967	1.5	30.8	.04	1,040.	1.6
100	1,289	2.0	40.9	.06	1,389.	2.2
500	6,446	10.1	204.	.31	6,944.	10.8
1,000	12,890	20.1	409.	.62	13,890.	21.7

1/ Application rate of 2 inches per week

2/ Application rate of 1 foot per day for 10 days then dry for 5 days

3/ Average application same as Campbell Soup, Paris, Texas, 0.2 inches per day

CHAPTER III - HYDROLOGY OF NATURAL SYSTEMS

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Application of substantial quantities of waste water to large contiguous
areas requires an evaluation of impacts on natural hydrologic regimes.
Climate, vegetation, soils and geology all interact to produce an inte-
grated end effect on naturally flowing hydrologic systems. These factors
are sufficiently diverse to require consideration of the five study areas
in three major groupings: California, the three Lake States areas and
New England. The following section will interpret the natural flow re-
gimes in relation to climate by evaluating water balance and the various
interactions influencing the hydrologic regimes.

In any area the significance of the influence of added waste water will be
in proportion to the area in the drainage basin affected. Small watersheds
which are 100% treated will be very significantly influenced by waste
water additions. As the small drainage contributes to successively larger
streams, impacts diminish. Each system of waste disposal will add an
equal amount of total water to a hydrologic regime, the differences being
spray irrigation and overland runoff apply the water at reduced rates over
much larger areas of the watershed. Rapid infiltration ponds concentrate
the point of application of waste but would apply an equal amount for an
equal population.

A. Flow Regime of Natural Systems

Surface Water Supply Papers of the United States published by the U.S.
Geological Survey were used to prepare summaries of the flows of near natural
systems in each of the study areas.

The California Study Area Flow Regimes: Table III-1 shows the total monthly
discharge (cfs-days and inches) for three representative small watersheds in
the Sacramento area. Bear Creek is a hundred square mile watershed drain-
ing the east front of the coastal range. Flow patterns typically follow
rainfall patterns. The winter months have high flows while the summer have

Table III-1

MONTHLY FLOW OF SELECTED RIVERS IN THE CALIFORNIA AREA

	Bear Creek (4517.2) *		Dry Creek (4473)			Arcade Creek (4473.6)	
Area Sq. Mi.	100		0.39			31.5	
	Total Flow		Total Flow		Rainfall	Total Flow	
	cfs-days	inches	cfs-days	inches	(inches)	cfs-days	inches
October 31	42.9	.12	0.3	.03	0	183.1	.21
November 30	65.1	.17	1.1	.10	0.8	377.7	.44
December 31	593.4	1.60	29.0	2.77	5.1	2,748.8	3.25
January 31	148.6	.40	21.8	2.08	3.2	1,731.1	2.04
February 28	6,748.3	18.10	.1	.01	.4	49.5	.06
March 31	3,284.0	8.83	0	--	1.1	156.5	.18
April 30	435.6	1.15	4.0	3.8	2.9	842.0	1.00
May 31	218.2	.59	0	--	.1	19.9	.02
June 30	70.1	.19	0	--	0	58.0	.07
July 31	46.6	.12	.2	.02	0	77.1	.09
August 31	30.3	.08	.3	.03	.5	62.3	.07
September 30	36.0	.10	.1	.01	0	421.2	.50

Mean Yearly cfs

1960-61	13.5	--		--	--
1961-62	32.1	--		--	--
1962-63	53.3	--		--	--
1963-64	9.4	.084		11.0	7.71
1964-65	60.6	.16		14.1	18.4
Average	30.1				

*Bear Creek is identified by 4517.2 in the U.S. Geological Survey Records

very low flows. The mean annual flow in cubic feet per second (cfs) varies tremendously between water years as shown by the change from 9.4 cfs in 1963-64 to 60.6 cfs in 1964-65; the average is 30.1 cfs.

Dry Creek and Arcade Creek drain smaller areas west of the Sacramento River. In the case of Dry Creek a precipitation station is adjacent to the gaging station. The watershed area of 0.39 square miles is not sufficient to sustain annual flow. Again, the flow regime very definitely follows a pattern of rainfall with extended periods of zero flow during rainless months. The station has recently been established and shows a variation in mean annual average discharge from 0.084 cfs in 1963-64 to 0.160 cfs in 1964-65. In these two years rainfall was 11 and 14.1 inches, respectively.

Arcade is a 31.5 square mile watershed just south of Dry Creek. Arcade Creek is eighty times larger than Dry Creek and shows a corresponding approximate eighty-fold increase in average daily streamflow. With appropriate changes between water year 1963-64 (7.71 cfs) and water year 1964-65 (18.4 cfs) monthly changes in total flow do not reflect this same relationship, however, as maximum winter flows are 500-fold greater in Arcade Creek than Dry Creek.

The Lake States and New England Flow Regimes: Flow regimes of the Lake States and New England areas are sufficiently similar to be considered under one heading. Table III-2 shows monthly flows of one river in the Lake States area and two in New England. The Popple River is a 131 square mile watershed tributary to Lake Michigan. Runoff for two water years varied from 9.11 inches in water year 1963-64 to 14.06 inches in water year 1964-65. Major effects of climate on flow regimes relate to spring snowmelt in both the Lake States and New England. Streamflow increases due to snowmelt in the spring months. Lowest flows occur in late summer increasing in early winter with rain, then decreasing as snow accumulates in January through early March.

The rivers used in Table III-2 are relatively unregulated, so flow regimes reflect natural discharge rates. Flow values are given for the 1963-64 water year and compared with recorded data for previous water years where possible.

- 14 -

Table III-2

MONTHLY FLOW OF SELECTED RIVERS IN THE LAKE STATES AND NEW ENGLAND STUDY AREAS FOR WATER YEAR 1963-64

	Popple River (4-0637)		Nashua River (1-0965)	Assabet River (1-0970)
Area Sq. Mi.	131		316	116
	Total Flow			
	cfs-days	inches	Inches	Inches
October	1,362	0.39	0.44	0.23
November	1,486	.42	1.13	1.18
December	1,224	.35	1.35	1.66
January	1,000	.28	2.17	2.40
February	766	.22	1.77	2.18
March	944	.27	4.20	4.35
April	5,728	1.63	3.40	3.42
May	8,885	2.52	1.22	1.27
June	1,639	.47	.52	.38
July	907	.26	.54	.32
August	3,686	1.10	.27	.16
September	4,286	1.21	.32	.12

Mean Daily cfs and Total Inches of Runoff for Water Years

1960-61	--	--	25.75	--
1961-62	--	--	21.05	21.48
1962-63	--	--	23.18	23.43
1963-64	87.7	9.11	17.33	17.66
1964-65	136.	14.06	8.83	9.34

Considerable variation exists in mean annual flow between water years as shown by the Nashua River; 8.83 cfs in 1964-65 to 25.75 in 1960-61. The Assabet River though smaller in total area shows a very similar seasonal flow regime.

B. Climate

Climate of the five study areas may be grouped in three general categories: Again, California, the Lake States study areas and New England.

The California Study Area: The long time weather records at Davis Agricultural Experiment Station have been used to characterize the climate of the California study area. The climate is typically maritime with major amounts (70 percent) of precipitation occurring as rain during the wet winter months, December through March. Temperatures seldom drop below freezing in the lower elevations, thus, precipitation occurs as rain with soil temperatures consistently above freezing. Long time average climate is summarized in Table III-3 showing mean monthly temperatures, rainfall, evaporation and soil temperature. Evaporation at Folsom Dam and Ferndale is compared with Davis to show local variation.

An additional characteristic of climate which must be considered is the potential for maximum amounts of rainfall. Table III-4 shows the maximum recorded 24 hour rainfall by months for both Sacramento and Davis, California. With the exception of the dry summer months, June, July and August, both of these stations report the potential for large amounts of daily precipitation. Sacramento may receive up to three inches of rain per day between December and April. Davis has the highest 24 hour rainfall (4.12 inches) and may receive an excess of an inch and a half per day between October and May.

The Lake States Study Area: Average temperature, precipitation, evaporation and soil temperatures at four inches have been used to characterize the general climate of the Lake States study area. Urbana, Illinois and Wooster, Ohio Agricultural Experiment Station data have been used in Table III-5. Mean average daily temperatures drop below freezing for 3 months, December,

- 16 -

Table III-3

MEAN MONTHLY AIR TEMPERATURE, PRECIPITATION, EVAPORATION AND SOIL TEMPERATURE AT FOUR INCHES FOR DAVIS, CALIFORNIA WITH EVAPORATION AT FOLSOM DAM AND FERNDALE

	Jan.	Feb.	Mar.	Apr.	May	Jun.	Jul.	Aug.	Sept.	Oct.	Nov.	Dec.	Annual
DAVIS, CALIFORNIA													
Air Temperature (°F)	45.3	49.3	53.4	58.9	64.7	70.9	75.2	73.3	71.1	63.2	53.1	46.3	60.4
Precipitation (inches)	3.29	3.25	2.14	1.36	.57	.13	0	0	.17	.76	1.37	3.41	16.46
Evaporation (inches)	1.29	1.96	3.90	5.89	8.38	10.13	11.05	9.91	8.10	5.26	2.55	1.40	79.82
Soil Temperature (°F)	43	47	55	64	77	81	88	88	82	65	57	49	
FOLSOM DAM EAST OF DAVIS													
Evaporation (inches)	.4	.9	3.5	5.1	8.8	8.6	11.4	10.8	8.3	4.5	2.1	.8	65
FERNDALE ON COAST													
Evaporation (inches)	.5	.8	2.7	3.2	3.8	3.2	4.7	4.6	3.6	2.3	1.0	.6	31

Table III-4

MAXIMUM DAILY RAINFALL (INCHES) FOR THE THREE STUDY AREAS

| | California | | Lake States | | New England | |
	Sacramento	Davis	Urbana Ill.	Wooster Ohio	Mass.	Merrimack N.H.
January	3.52	4.12	2.86	1.56	1.36	1.69
February	3.29	2.42	1.56	1.50	1.29	1.54
March	2.52	1.66	2.17	1.52	2.28	1.57
April	3.35	1.22	2.87	1.85	2.22	1.87
May	1.93	1.50	3.28	1.9?	1.99	2.02
June	.60	.52	2.27	3.44	3.06	2.62
July	.07	.01	2.37	2.31	2.25	3.34
August	.01	.05	3.44	2.10	2.61	2.40
September	3.14	.35	2.08	1.80	3.15	3.94
October	1.44	1.54	2.88	1.70	2.29	3.94
November	2.13	1.91	1.35	1.45	2.88	3.10
December	2.99	1.87	2.29	1.24	2.37	3.15

Table III-5

MEAN MONTHLY AIR TEMPERATURE, PRECIPITATION, EVAPORATION AND SOIL TEMPERATURE AT FOUR INCHES FOR URBANA, ILL. AND WOOSTER, OHIO

	Jan.	Feb.	Mar.	Apr.	May	Jun.	Jul.	Aug.	Sept.	Oct.	Nov.	Dec.	Annual
URBANA, ILLINOIS													
Average Temperature (°F)	28.7	31.0	39.5	51.9	62.5	72.4	76.2	74.4	67.3	56.6	41.3	31.1	52.7
Precipitation (inches)	2.16	2.09	3.17	3.54	3.22	4.54	3.49	3.04	3.04	3.01	2.62	2.08	37.0
Evaporation (inches)				3.79	5.37	6.32	6.52	5.98	4.70	3.24			42.0*
Soil Temperature (°F)	29	32	38	52	62	70	79	76	69	57	44	34	53.3
WOOSTER, OHIO													
Average Temperature (°F)	28.3	29.0	36.7	50.8	58.6	68.3	72.0	70.5	63.6	52.8	40.1	30.1	49.8
Precipitation (inches)	2.80	2.11	3.0	3.19	3.85	4.18	4.97	3.61	2.84	2.21	2.14	2.21	36.21
Evaporation (inches)				3.83	5.15	6.18	6.71	5.86	4.51	2.68			41.0*
Soil Temperature (°F)	31	32	34	47	58	64	72	71	63	54	42	34	50.1

*Evaporation reported for growing season only; total annual value is estimated

January and February, in the Lake States area. Soil temperatures at the
four inch depth are consistently below freezing for only January and February
in Urbana and only January in Wooster. Precipitation is typically continental
with maximum monthly rainfall occurring during the summer growing season and
minimum quantities during the winter months. At both stations annual eva-
poration is estimated to exceed precipitation by approximately five inches.
Maximum deficits of two to four inches occur during May, June, July and
August.

The New England Study Area: The Merrimack Basin climate is colder and sea-
sonally wetter than the other study areas. The closest station reporting
soil temperature and evaporation (Kingston, R.I.) is warmer and drier than
the northern headwaters of the basin. Snow is usually persistent from
early December through late February, averaging 50 days in lower portions
of the watershed and 130 days in the northern headwaters. Freezing tempera-
tures prevail from mid-December through February resulting in frozen soils
(four inches) in January, February and early March. (See Table III-6)

Maximum potential rainfall rates for 24 hour periods are shown in Table III-4.
In contrast to the California study area, the potential for major storms is
of prime concern during the summer growing season. The Lake States have a
potential for rainfall in excess of three inches per day from April through
August. The New England area has a potential for maximum rainfall rates in
excess of three inches per day from June through December.

C. Water Balance

The integrated influences of all factors reducing the yield of water from
a watershed may be balanced against those factors contributing to water in-
puts and considered in terms of a water balance. An annual water balance
for any stream considers total annual water input in relation to total
annual streamflow, the deficit being actual water losses. Though difficult
to assess accurately it is convenient to consider these losses in terms of
inches of depth per unit land area. An explanation of terms and processes
would be appropriate before considering water balance.

Table III-6

MEAN MONTHLY AIR TEMPERATURE, PRECIPITATION, EVAPORATION AND SOIL TEMPERATURE AT FOUR INCHES FOR KINGSTON, R.I., WITH EVAPORATION FOR ROCHESTER, MASS.

	Jan.	Feb.	Mar.	Apr.	May	Jun.	Jul.	Aug.	Sept.	Oct.	Nov.	Dec.	Annual
KINGSTON, RHODE ISLAND													
Temperature (°F)	29.2	29.8	36.4	45.8	55.6	64.3	68.1	69.1	62.2	52.4	42.5	31.8	49.1
Precipitation (inches)	2.31	4.10	3.31	3.89	3.05	1.60	4.05	3.79	4.38	2.83	7.13	10.28	50.72
Evaporation (inches)					6.01	6.28	4.81	5.22	3.27	2.32			35.0*
Soil Temperature (°F)	30.8	30.4	31.8	45.2	55.1	65.8	68.5	68.7	67.8	58.1	47.6	37.6	50.6
ROCHESTER, MASSACHUSETTS													
Evaporation (inches)				3.76	4.48	5.13	4.39	5.36	3.15	2.28			30.0*

*Evaporation reported for growing season only; annual total estimated

Evaporation is the conversion of liquid water to the vapor phase requiring a sustained source of energy (heat) for the process to be continuous. Evaporation takes place from free water surfaces and liquid water on the surfaces of soils or plants. Evapotranspiration is the combined term of evaporation from free water surfaces along with evaporation of stored soil moisture through plant systems by the transpiration process. Frequently, it is very difficult to separate evaporation losses as they occur through evaporation of intercepted water from losses such as transpiration.

The potential quantity of energy available for evaporation of water is termed potential evapotranspiration. Usually, living green plants are very efficient absorbers of infrared radiation (long wave or heat source). A very high percentage of available heat is used for transpiration of water from plant systems. When stored soil moisture is depleted, plant mechanisms restrict water loss thus, plants will not transpire at the potential rate.

Several methods are available to estimate evapotranspiration and potential evapotranspiration. Very frequently, potential evapotranspiration can be indexed by evaporation from a free water surface as measured in the standard 4-foot Class A Weather Bureau Evaporation Pan. This study has considered the data reported by the Weather Bureau as evaporation loss in inches per month to approximate the potential evapotranspiration losses. It has been demonstrated that water losses in irrigated areas perform approximately as evaporation from Class A pans. These data are frequently used to establish irrigation schedules.

The water balance of a watershed represents the balance between total rainfall input (R) and streamflow output (F). The differences being an evapotranspiration loss (ET), deep seepage, or a change in basin storage capacity (S):

$$R = F \pm ET \pm S$$

This simple equation assumes that water measured as streamflow represents the bulk of the water output of a basin. If substantial deep seepage occurs

- 22 -

or losses in deep aquifers, these factors should be considered in the
water balance on a larger regional scope.

Evapotranspiration losses can be further subdivided into interception loss
in the vegetative canopy (Ic), interception loss in decaying litter layers
(If), transpiration loss (T) due to water used by plants and evaporation
from soil (Es) or free water surfaces (Ew):

$$ET = Ic + If + T + Es + Ew$$

Native forest and brush vegetation would have high water losses due to inter-
ception by the aerial portions of plants and litter. These native plants
often completely utilize stored soil moisture by transpiration during the
summer dry months. By comparison, irrigated crops would have high evapora-
tion loss from wet soil surfaces and free water on plant surfaces in addition
to the transpiration loss. The balance in water losses are important in
evaluating the impacts of waste water disposal schemes on flow regimes in
each study area. A general water balance will be considered for each area
based on recorded climatological data (Tables III-3,5,6).

The California Study Area: Mean annual precipitation varies from 12 to
about 30 inches in the study area with variations of 30 to 80 inches in
annual potential evaporation loss. Only streams with their headwaters in
higher mountain ranges sustain streamflow in dry summer months. The flow
regime of Dry Creek (Table III-1) is a usual example of water balance for
the low foothills.

Maximum monthly evapotranspiration occurs in June and July following patterns
of maximum temperature and day length. At Davis, California potential eva-
poration exceeds precipitation from March through November (Table III-3).
Maximum potential monthly evaporation is 8 to 11 inches during summer months
of no rainfall. Figure III-1 shows the deficit of potential evapotranspira-
tion in relation to precipitation.

The deepest soils with large amounts of stored moisture usually have about
12 inches of available water which is completely utilized by plants in tran-
spiration by early May. Only very deep rooted plants will continue to tran-
spire. Winter rains recharge the soil storage capacity, leaving very little

- 23 -

water for surface runoff in streams. Higher intensity storms contribute directly to overland flow and streamflow. Figure III-1 shows only relative relations as data are not available for all parameters on a given watershed.

The Lake States Study Area: Average annual runoff (15 inches) is approximately 40% of the average annual precipitation (37 inches). Twenty-two inches are lost by evapotranspiration discounting changes in soil moisture and basin moisture storage characteristics.

Figure III-2 shows maximum seasonal rainfall occurs during months of maximum potential evapotranspiration. Increased streamflow occurs as snow, accumulated during the winter, melts with warm spring weather and increasing rain. Maximum streamflow occurs before maximum precipitation falls as increasing evapotranspiration loss rapidly reduces moisture available for streamflow. Maximum evapotranspiration losses cause minimum streamflow even during months of maximum rainfall. An unusually wet August and September caused some departure from the expected average hydrograph (Figure III-2).

The moisture deficit (potential evapotranspiration minus rainfall) during the growing season can be supplied by stored soil moisture on deeper soils. Under this situation actual water use by plants will almost equal the potential loss.

The New England Area: Water balance of the New England study area is characterized by less potential evapotranspiration and increased annual precipitation (Figure III-3). Monthly precipitation occurs in maximum quantities in early winter and early spring. Late growing season (September) rainfall is also adequate for potential evaporation. Evaporation potential exceeds rainfall in May, June, July and August, however, the annual water balance yields about 15-25 inches more water available for streamflow than for evaporation loss.

Streamflow in both the Lake States and New England shows an expected increase with spring snowmelt and lowest flows during seasonal maximum evapo-

- 24 -

transpiration periods. Soil moisture is probably deficient in July and
August reducing actual water losses below potential losses. The average
annual balance shows about 50% of precipitation is lost by evapotranspira-
tion (25 inches) and 50% occurring as streamflow (25 inches).

ANNUAL WATER BALANCE FOR SELECTED STATIONS IN THE CALIFORNIA AREA
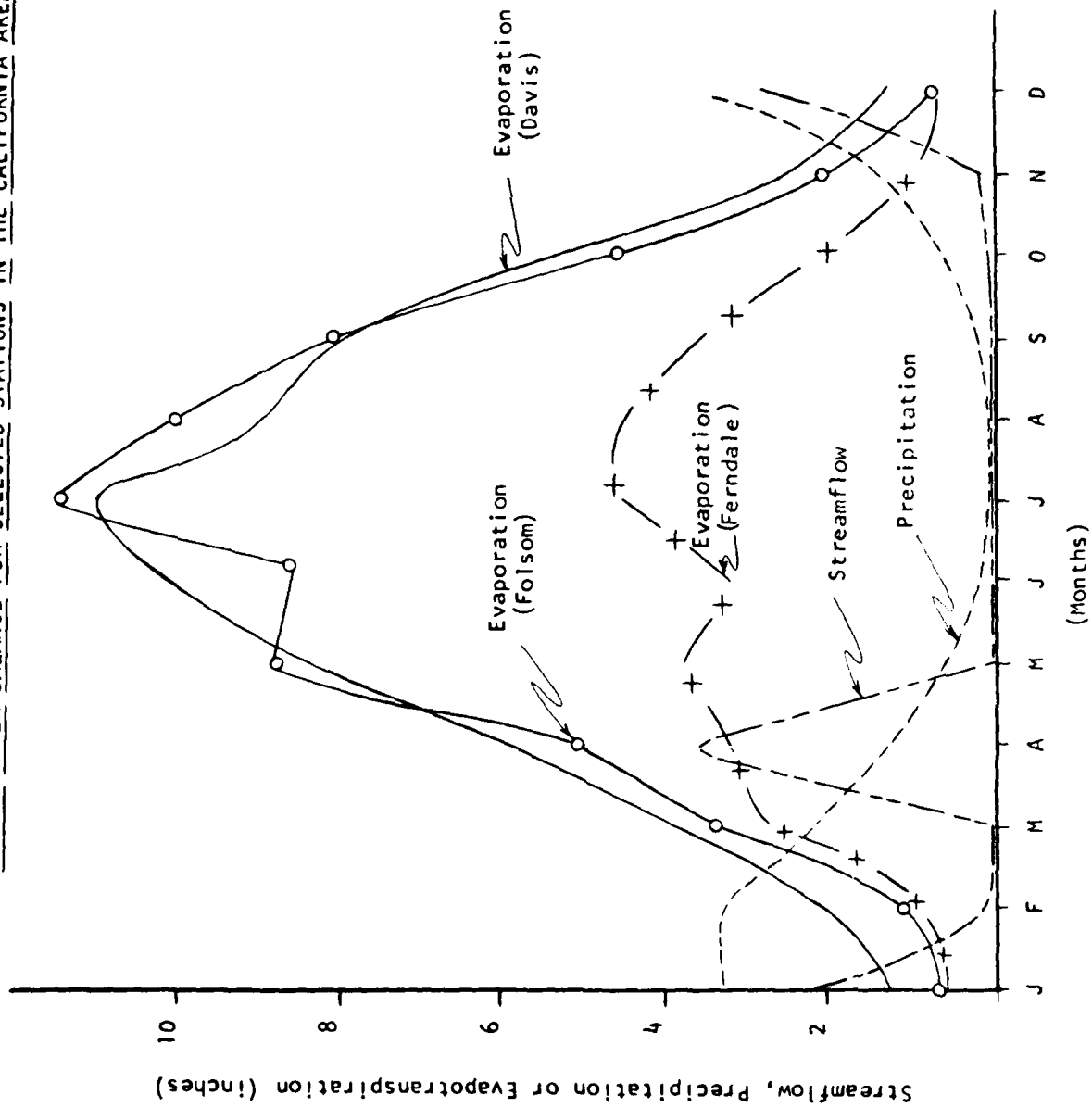
Figure III-1

Figure III-2

ANNUAL WATER BALANCE FOR SELECTED STATIONS IN THE LAKE STATES AREA

Wooster, Ohio

Figure III-3

ANNUAL WATER BALANCE FOR SELECTED STATIONS IN THE NEW ENGLAND AREA

Precipitation (Kingston)

(Kingston)
Evaporation (Rochester)

Streamflow

(Months)

Precipitation or Evaporation (Inches)
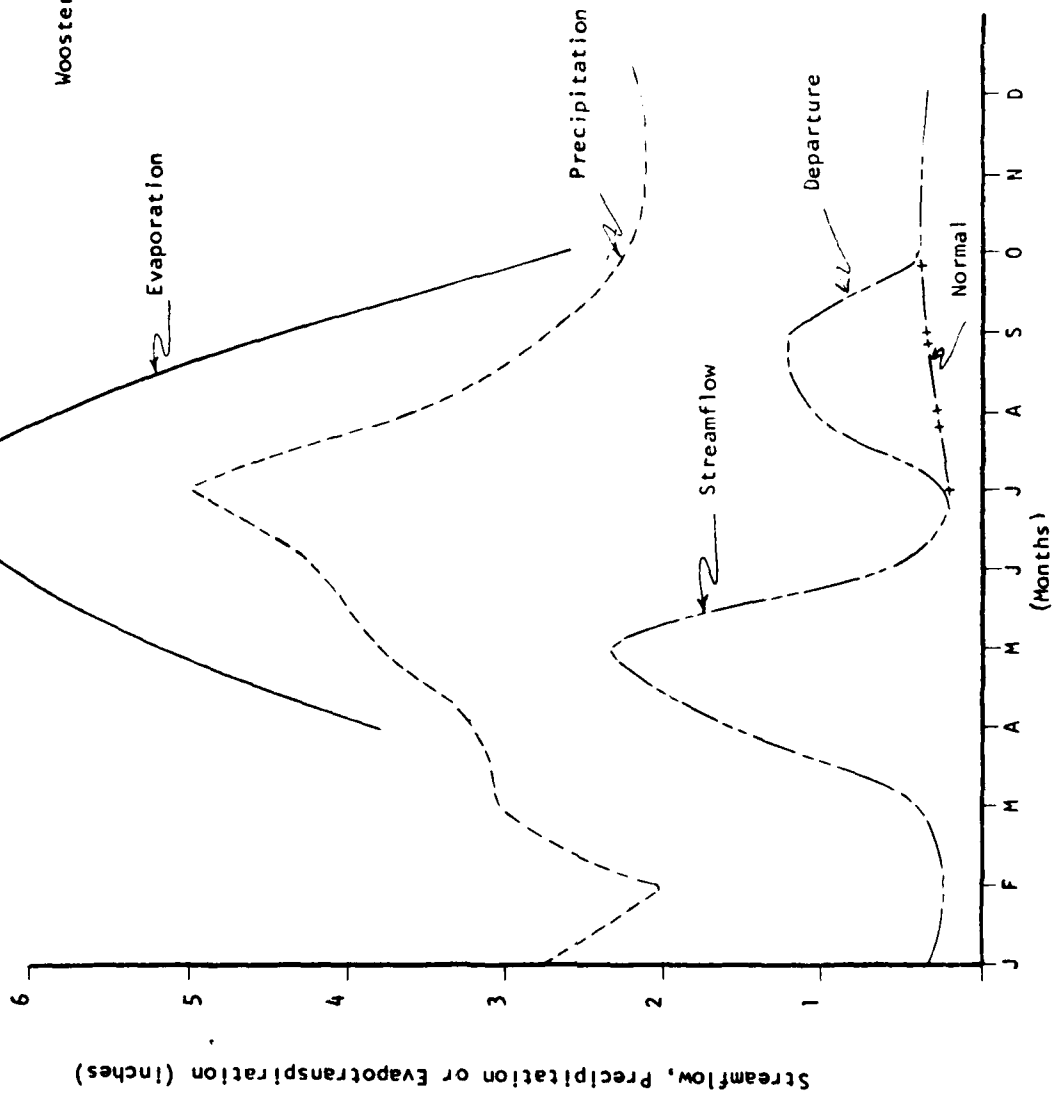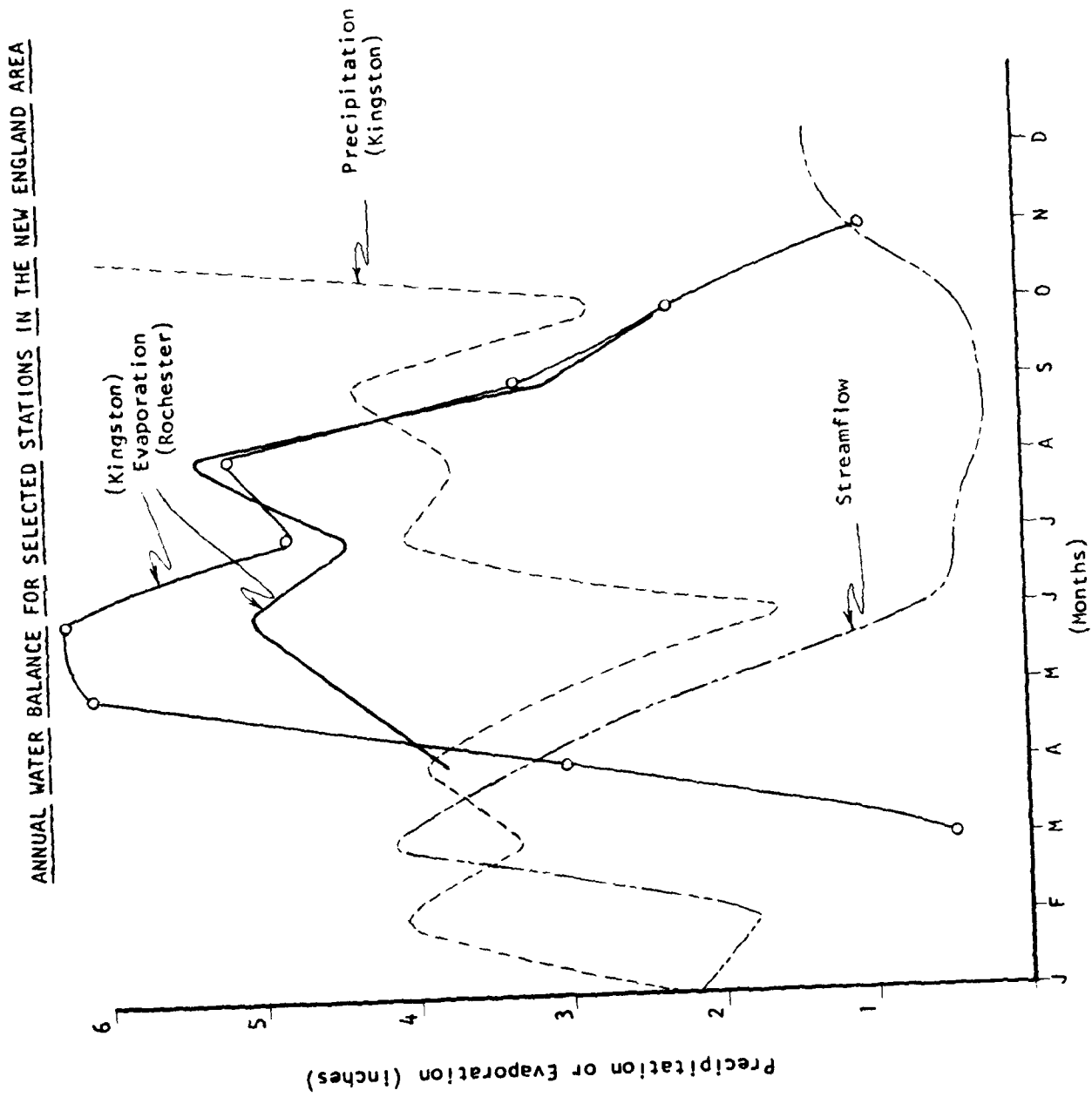
CHAPTER IV - WATER QUALITY OF NATURAL ECOSYSTEMS

Water quality of streams as identified by dissolved and suspended material is in equilibrium with climate, vegetation, soils and geology for natural or near natural ecosystems. Major impacts of man's activities on the natural systems usually are related to altering the vegetation from prairie or forest to agricultural or urban uses. These impacts may result in marked deviations in water quality. Management of areas for agricultural uses frequently includes application of many types of chemicals. Again, certain of these chemicals are responsible for significant changes in the water quality of streams.

A.   Concepts of Ecosystem Nutrient Cycling

Any consideration of nutrient cycling must recognize water as the universal solvent and transport agent for nutrients or other chemicals in the ecosystem. The ecosystem can generally be considered as five major components: Atmosphere, vegetation, dead organic materials, soils, and geologic parent materials.

The atmosphere is the source of many chemicals and nutrients particularly, carbon, hydrogen, oxygen and nitrogen required for plant growth. The atmosphere, as described by long term weather patterns, yields a particular climate for a given spacial situation thus providing changes in moisture regime as well as temperature which interact with other components to maintain a given natural system.

The natural systems occurring in the five study areas exhibit a broad range of characteristics. Much of the study area in California is in a grass-chaparral zone too dry for even the Ponderosa Pine forests. The total biomass ranges from 1-5 tons per acre primarily in the combined forms of carbon, hydrogen and oxygen which make up cellulose, lignins and related plant materials. Many annual plants occur in arid zones thus, cycling of certain nutrients is quite rapid. Organic matter accumulations as litter are not usually large thus, relatively small amounts of organic matter occur in soils.

- 29 -

Vegetation of both the Lake States and New England was originally forest;
oak hickory forests occurring through the southern Great Lakes region and
birch, beech, maple and hemlock forest in the New England area. Forest
trees by virtue of their longevity and woody structure accumulate tremen-
dous amounts of biomass (from a few tons to several hundred tons per acre
depending on age and species). In addition forest litter may accumulate
in amounts exceeding 150 tons per acre. In both hardwood and evergreen
forests there is a dynamic exchange of several tons of leaf and needle
fall a year between the crowns of the trees and the forest soil. Although
the systems have   dynamic internal cycling they frequently establish
an equilibrium in terms of inputs to an ecosystem watershed and outputs
in streamflow.

Vegetation provides a varying area of live surface which interacts with
climate and influences soil to produce organic material through the pro-
cess of photosynthesis. This process fixes carbon, hydrogen, oxygen and
nitrogen making these elements available in the continuing cycle.

The dynamic plant surfaces also intercept large quantities of precipita-
tion both as rain and snow evaporating 10-15% of the annual precipitation
back to the atmosphere. The surface area of vegetation also gives an in-
dex to the potential for transpiration loss. Plants with high amounts of
reactive surface area have a high capacity for transpiring water. Forest
trees usually have very large surface leaf or needle areas, ranging from
three to twelve times the crown projection area.

Cycling of both plant and animal organic matter is an extremely important
phase of ecosystem dynamics as organic matter is the energy source for all
decomposers in the system. Organic materials provide an insulation of the
soil from the atmosphere so that moisture and temperature conditions become
much more stable. In addition it provides a mechanical protection which
prevents erosion and clogging of soil pores.

The soil provides a mechanical anchorage for plants as well as a supply of
water and nutrients necessary for plant life processes. The ability of any

soil to provide both chemicals and water to plants is related to the
active fraction of the soil as indexed by surface area. Soils of coarse
texture with large gravel and stone contents have very little surface area, thus
very little capacity to supply nutrients or water. At the other extreme,
soils with very small particles (clays) have a very high capacity for
storage of nutrients and water but frequently may have poor physical
properties for growth of plant-root systems.

The interaction between living plants and animals, organic matter, climate
and parent geologic material supply ingredients for the formation of soils.
Forest soils are normally acid ranging from pH 3.5 to near neutral. Or-
ganic acids leaching through the forest litter layer transport base nutri-
ents deeper in the soil. This leaching process also provides the weather-
ing phenomena for decomposition of parent geologic materials in a complex
chemical reaction.

The concept of ecosystem nutrient and chemical cycling suggests inputs to
the system from the atmosphere by precipitation or fixation, weathering
of parent material or man-induced inputs. Losses of nutrients and chemicals
from an ecosystem may occur as dissolved and suspended materials in streams,
deep seepage and groundwater, by transfer or removal of plant material or
by release of fixed materials to the atmosphere through volatization.

B. Limiting Factors in Ion Movement in Ecosystems

Ecosystems in humid and temperate regions are known to be dynamic. Commonly
several tons of air-dry material will be produced annually to either accumu-
late in the biomass or return to the soil surface for decomposition.

While internal processes of ecosystem function are extremely dynamic the
overall chemical balance on a watershed basis is frequently almost static.
Atmosphere inputs of chemicals is in near equilibrium with output of these
same chemicals in streamflow. These general conditions have been found to
prevail even in relatively high precipitation zones in the foothills of the
Cascade Mountains in Washington.

- 31 -

Semi-natural conditions where vegetation, soil and climate are in equilibrium with biological and soil formation processes will have a consistent ion movement in soils and streams. Disruption of this equilibrium requires a dynamic change in some phase of the system. This may be accomplished by altering biological processes through vegetation manipulation or adding substantial quantities of ions which alter processes and soil chemical equilibria.

Review of literature related to this field definitely suggests that limiting factors in ion movement in ecosystems are not related to precipitation but rather related to chemical inputs.

It has frequently been demonstrated that drastic alteration of watershed conditions which furnish a large input of chemicals to the soil will result in a substantial output of chemicals to streamflow. Fredrickson (1971) reports marked increases in bicarbonate and nitrate discharge following logging and slash burning of small forested watersheds.

Pearson and Fischer (1971) have shown that the quantities of base elements (calcium, magnesium, sodium and potassium) in unpolluted streams in New England are substantially greater than the inputs due to precipitation. Anions of sulfate and chlorine are approximately equal in streamflow and precipitation. Nitrogen in the form of NO_3 is the only element conserved by soil systems occurring in lesser concentrations in streamflow than in precipitation. Bormann, et al (1967) have shown 3 to 20 fold increases in cation (Ca, Mg, Na and K) loss and reversal of conservation of nitrate following forest cutting.

We may expect that the application of up to 100 inches of waste water annually with model constituents would alter processes and the equilibrium of chemical movement in any ecosystem. Many of the constituents would be retained by the soil while other natural constituents would be displaced and contributed to groundwater and eventually surface water.

After manipulation or with sustained additions of waste water, eco-
systems will reach a new equilibrium state providing vegetation remains
stable (no disastrous fires) and weather patterns are near average.
Water quality as measured by surface water draining a particular water-
shed will represent a new chemical equilibrium where decomposition of
plant material and weathering of soil yields chemicals through the soil
solution to streams in balance with new rates of biological and chemical
processes.

C. Precipitation Chemistry

Chemical composition of precipitation in the eastern Lake States and New
England was reported by Pearson and Fischer (1971) in a one to three year
study. They found that the loads of certain elements did not vary sys-
tematically either with area or annual precipitation. These elements
are reported in tons per day per square mile as calcium - 5.3 X 10^{-3},
magnesium - 1 X 10^{-3}, potassium - 1.4 X 10^{-3}, and total nitrogen - 2.2 X
10^{-3}, sodium and chlorine decrease with distance from coastal stations.
Hydrogen ions and sulfate loads vary with precipitation in both inland
and coastal areas. These limited data suggest that higher calcium,
nitrate and sulfate loads may be expected near industrial areas.

The input of chemicals particularly forms of nitrogen also has been
studied for selected stations across the United States. Junge (1958)
shows seasonal concentrations in ammonia and nitrate in rainwater (Table
IV-1). Concentrations vary seasonally in each study area from insignifi-
cant amounts on both coasts to rather substantial amounts in the Lake
States area. Using these approximate values in the Pacific Northwest,
Cole (1965) has found an approximate equilibrium between the input of
total nitrogen in rainwater and outflow through the soil system.

Conversion of data in Table IV-1 for total nitrogen input by seasonal
rainfall patterns results in total nitrogen inputs which vary from 1 to
5 lbs./acre/year.

- 33 -

Table IV-1

CONCENTRATION OF AMMONIA AND NITRATE IN RAINWATER
IN THE STUDY AREAS. ALL VALUES IN MG/L

	California	Lake States	New England
Ammonia (NH_3)			
July - Sept 1956	Trace - 0.03	0.21 - 0.27	0.08 - 0.21
Oct - Dec 1956	0.01 - 0.07	0.02 - 0.05	0.01 - 0.03
Jan - Mar 1957	0.03 - 0.20	0.18 - 0.42	0.01 - 0.05
Apr - June 1957	0.50 - 1.75	0.07 - 0.27	0.03 - 0.07
Nitrate (NO_3)			
July - Sept 1956	0.50 - 1.50	0.86 - 1.36	0.30 - 1.00
Oct - Dec 1956	0.30 - 0.50	0.50 - 1.64	0.15 - 0.70
Jan - Mar 1957	0.23 - 0.50	1.25 - 1.76	0.05 - 0.30
Apr - June 1957	0.30 - 0.50	0.76 - 1.00	0.20 - 0.30

D. Dissolved Constituents in Streams

Many constituents dissolved in streams, Tables IV-2 and IV-3, having natural sources in the parent rock material interact with acidity, temperature, and soil formation processes.

Silica (SiO_2) is dissolved from practically solid rock to some extent. Natural surface water usually contains less than 10 mg/l of silica, but may contain up to 50 mg/l. Consistently higher values for silica are found for California rivers than New England.

Aluminum (Al) and iron (Fe) usually occur in fractions of a mg/l as both are relatively insoluble near neutral pH water. More acid waters may contain higher quantities of these elements as dissolved material.

Manganese (Mn) may be dissolved in appreciable quantities from certain geologic materials. In acid soils higher quantities of manganese are frequently associated with conditions which contribute to increased concentrations of aluminum and iron.

Calcium is dissolved from most rocks and soils but has highest concentration in zones particularly if soils are formed from Limestone, dolomite or gypsum. In these conditions calcium may range from 50 to 100 mg/l. In regions of higher rainfall particularly under acid soil formation conditions, associated with granitic soils, calcium is significantly reduced. This is illustrated by comparison of the quantity of calcium in the selected rivers of the California study area as compared with the quantities in the Lake States and New England (Tables IV-2,3).

Magnesium, like calcium, is dissolved from many rocks and soils but particularly from dolomitic rocks. The pattern for magnesium occurrence is very similar to calcium in the study areas.

Sodium (Na) and potassium (K) are dissolved from many soils and rocks and frequently are the predominant cations in highly mineralized water in arid

- 35 -

RANGE IN WATER QUALITY OF SELECTED NEW ENGLAND RIVERS AND TRIBUTARIES TO THE GREAT LAKES*

River	SiO2	Al	Fe	Mn	Ca	Mg	Na	K	HCO3	SO4	Cl	F	NO3	PO4	Spec. Cond.	pH	TDS*
Merrimack 1-1000 4425 mi²	2.9-6.9	0.1-0.8	0.14-0.42	0.04-0.07	4.2-13	0.6-1.7	2.0-19		5-34	9.7-18	7.2-22	0.1-0.4	0.4-6.7	0.16-1.11	62-146	5.6-6.8	47-107
Miller 1-1640 167 mi²	3.6-7.9		0.10-1.90	0.00-0.14	5.2-8.7	0.9-3.9	6.6-38	0.9-2.7	4-80	0.2-26	9-20	0.1-0.3	0.4-2.0		69-247	5.5-7.0	52-177
Lk. Michigan Popple River 4-0637					6.7-24	4.1-13	0.6-2.2	0.3-0.6	32-133	2.8-12	0.0-1.0	0.1-	0.6-1.2		72-213	6.2-7.1	88-146
Lk. Erie Maumee 4-1940.1							16-36		137-217	52-110	15-40		0.3-21.0	0.56-0.88	412-688	7.0-7.9	311-450
Sandusky 4-1980			0.07-0.46	0.00-1.80					82-268	53-258	12-44	0.2-1.1	1.8-37	0.38-1.80	328-978	7.2-8.2	233-666

* All values are in mg/l except pH and specific conductance (micromhos per centimeter at 25°C.

TableIV-2

WATER QUALITY OF SELECTED RIVERS IN THE CALIFORNIA STUDY AREA

Table IV-3

River	Date	Area Sq. Mi.	SiO$_2$	Fe	Ca	Mg	Na	K	HCO$_3$	CO$_3$	SO$_4$	C$_1$	F	NO$_3$	B	Spec. Cond.	pH	Total Dissolved
Cache	5/ 4/66	198	17		28	24	22	1.1	191	9	13	23		0.8	1.6	404	8.5	236
Cache	9/14/66		21		40	30	39	2.0	236	0	21	72		2.1	4.1	613	8.1	347
Putah	5/20/66	547	13		20	23	8.3	1.3	170	2	17	4.7		0.3	0.1	301	8.3	174
Putah	9/ 6/66		13		17	25	8.2	1.5	170	0	17	4.0						
Bear	5/ 5/66	292	11		15	8	4.5	0.7	70	0	15	3.5		0.4	0	157	7.9	
Feather	5/ 5/66	5,921	11		7.4	2.6	2.5	0.7	36	0	3.0	.6		0.4	0	70	7.6	45
Sacramento	6/ 8/66	24,000	18	0	17	7.2	15	1.0	89	2	15	10	0.2	1.9	0	208	8.3	131
Sacramento	8/19/66		16	0.02	13	6.7	12	1.1	80	0	10	6.7	0.1	1.4	0	172	8.0	106
Sacramento	9/12/66		19	0	16	11	19	1.2	118	0	13	14	0.2	1.5	.1	258	7.8	153
Napa	5/ 6/66	81.4	35		23	9.6	23	2.5	116	0	17	21		7.4	.7	303	8.0	198
Napa	9/16/66		26		33	18	18	2.2	186	0	17	12		8.0	.3	373	7.4	226

portions of the western United States. Sodium also occurs in larger
portions in coastal areas due to proximity to salt water.

Bicarbonate (HCO_3) occurs in significant quantities in most streams
due to respiration in soil systems and release of carbon dioxide by
soil organisms. Frequently, it is the dominant anion associated with
base cations.

Sulfate (SO_4) may occur as dissolved constituents from certain rocks
particularly in sedimentary materials in arid areas. It is also highly
associated with industrialization and is a more dominant factor in the
Great Lakes tributaries than effluent streams from the California area
rivers.

Chloride (Cl) and fluoride (F) dissolve from rocks and soil and are
contributed from atmospheric precipitation related to industrialization.
In humid regions chloride concentrations are usually quite low where as
in arid and semi-arid regions much higher concentrations of chloride
occur in streams. Chlorides are abundant and very soluble in most forms.

Nitrate (NO_3) in water is the final oxidation product of nitrogenous
materials which must have an organic source. Under natural conditions,
nitrate is present as fractions of a mg/l and frequently is a limiting
element in the productive capacity of natural ecosystems. Quantities
of nitrate in excess of a few (2-5) mg/l represent substantial pollution
due to leaching of fertilizer, sewage or industrial waste discharge.

Phosphate (PO_4) like nitrate has its main source in organic materials.
Natural ecosystems leach extremely small quantities (fractions of a mg/l)
into stream systems. Higher quantities of phosphate might have their
source in domestic or industrial sewage containing forms of detergent.

Boron (B) usually occurs in fractions of a mg/l but in some cases, it
may be substantially more. Many plants are boron sensitive and will not

- 38 -

tolerate boron concentrations in water in excess of one to two mg/l.
Boron is seldom found in appreciable quantities in acid soils of the
Lake States or New England but does occur in arid zones in the West.
No explanation was given for the high concentration of boron reported
for the Cache River in the California area (4.1 mg/l) (Table IV-3).

Water quality may be represented by average values of selected streams
in the study areas. In the mountainous West there is a very high
correlation between elevational zone of the water contributing area and
water quality. Streams with their headwaters in the snow zone contri-
bute relatively pure water (Feather River, 70 micromhos specific con-
ductance), while streams which integrate a wide elevational zone or
have their source at relatively low elevations have much higher loads
of total dissolved solids. Table IV-3 shows the Cache River with water
quality for monovalent and divalent bases which is not unlike the
quality of a secondary waste treatment effluent. Surface waters are
applied to land in irrigation as well as water pumped from wells. Well
water will have higher total dissolved solids. Quantities of water applied
in the usual irrigation practices in the California study area vary
annually depending on climate but range from 18 to over 170 inches of
water applied annually. If the average total dissolved solids is 150
mg/l, then these water application rates result in yearly application of
total dissolved solids from 1-8 tons per year through application of
irrigation water. The elemental composition of the total dissolved
solids may be inferred from Table IV-3.

The range of water qualities of selected rivers tributary to the Great
Lakes and in New England reflects wide variations (Table IV-2) due to
streamflow with day-to-day variations which must reflect pollution loadings
due to industrial waste. Rivers tributary to the Great Lakes are, in
particular, polluted with very high quantities of sulfate, nitrate and
phosphate. Base cations are considerably lower than the more arid California
river systems particularly in the New England area where rainfall exceeds
evapotranspiration losses.

- 39 -

The quality of water in all three study areas at times would be enhanced by direct application of the model effluent as concentrations of many elements are significantly less.

Maximum concentrations of total dissolved solids (which correlate closely with high values for specific conductance) usually occur during periods of minimum streamflow. Exceptions to this are high values for nitrate which frequently correlate with large quantities of suspended sediment.

CHAPTER V - SOIL PHYSICAL PROPERTIES RELATED TO WASTE WATER DISPOSAL

CHAPTER V - SOIL PHYSICAL PROPERTIES RELATED TO WASTE WATER DISPOSAL

Renovation of waste water applied to soils may be achieved by physical filtration, chemical combination, including chelation, and formation of complex organic molecules, volatilization, ion exchange, biological reaction and plant uptake. The three suggested methods of disposal, spray irrigation, rapid infiltration ponds and overland runoff, would each utilize combinations of the above to achieve renovation. Each method would in turn have special requirements related to the particular technique and site. While the prime emphasis is on disposal of waste water, other benefits might be gained from utilization of waste water and should be considered.

Physical properties of the soil important in renovation of waste water are related first to the physical retention of water so that the above renovation processes might be accomplished and second, sufficient aeration through the profile for final oxidation or chemical reaction. Soil physical properties important are the interaction of soil texture and soil structure. Soil texture is the distribution of primary particle sizes. Clay textured soils have a high percentage of primary particles two microns (0.002 mm) in size or smaller. Clay soils unless very well structured are very poorly aerated and have very low infiltration and percolation rates. The reactive surface area of a soil is a direct function of primary particle sizes. The smaller the primary particles, the greater the reactive surface area. Cation exchange capacity is a direct function of clay contents.

Silt textured soils have primary particles ranging in size from two microns to 20 microns (0.02 mm). Silt soils have a greatly reduced reactive surface area, but improved physical properties for transmission of water and air. Sandy textured soils are usually classified as fine sands (0.02 to .2 mm) and coarse sands (0.2 to 2 mm). Primary particles above 2 mm in size are classified as gravels. In a continuing order, reactive surface

area becomes greatly reduced with increasing primary particle size
through clay, silt, sand and gravel textures. In a hypothetical example
the surface area of a 1 cubic centimeter (cc) rock increases from six
square centimeters per centimeter for a gravel (1 cc) to 600,000 square
centimeters per centimeter for a colloidal clay size. In a like manner
a given weight or volume of 2 micron clays has a surface area 50 times
greater than that of a fine sand or 10 times greater than the same
weight of silt.

From this it becomes obvious that the physical reactive surface area of
a soil is important as the site chemical renovation. An ideal soil would
have a maximum reactive surface area and an adequate percolation rate with
sufficient aeration. These conditions may frequently be achieved in silt
textured soils with a small percentage of clay (3-6 percent) but suffi-
ciently high in organic matter to have well aggregated structure.

Soil structure is defined as the arrangement of soil particles. Primary
particles particularly of silt and clay are formed in secondary aggregate
through cementing by oxides, humus and other materials. The chemistry of
base saturation of clays will also influence soil aggregation and structure.

The simplest soil system for use in renovation of waste water would be
one that is topographically situated above any possible flooding by river
systems and has sufficient depth to groundwater for renovation by the
suggested disposal system.

Critical assessment of the ability of a particular soil to renovate secon-
dary treated waste water effluent depends first on infiltration rate and
second on sustained percolation rates. Establishment of infiltration
and percolation rates by soil tests have been extensively reviewed by
McGauhey and Krone (1967). They and others (Elazar, et al, 1971) have
reviewed the relationship between infiltration, percolation and soil
clogging. Two major factors are apparently involved in reductions in
infiltration and percolation rates due to waste water disposal. The first
factor relates to physical clogging of soil pores by suspended solid
materials washed in direct physical contact. The second factor relates

to formations of slimes due to anaerobic conditions (Chapter VII).
These factors will not be a problem in properly managed disposal systems.

Philipp (1971) reports the failure of a trickle system to dispose of
waste effluent from an insulation board mill. The effluent contained
5,400 mg/l of total solids in the form of wood fiber, starches and
certain bonding agents. A variety of factors caused failure in the
initial system. A redesigned system became functional in 1967 which
utilizes spray irrigation on an area seeded with reed canary grass.
The system achieves a 99% reduction in disposal of organic waste. The
author emphasizes the need for aerobic conditions for purification and
decomposition of the organic waste material. Sopper (1971) has also
reported the sustained capacity of a Hublersburg-Hagerston soil to accept
secondary waste effluent for an extended number of years when applied at
a rate of 2"/week to forest and an old field site.

Soil structural relations apparently have not been influenced by accumu-
lations of sodium on the exchange capacity (Sopper, 1971). In general
soil structure and aggregation is enhanced by alternate wetting and
drying cycles. Additions of organic matter are usually important in
maintaining stable soil structural relations. Soils initially very
low in organic matter will increase their organic matter content with
applications of waste water. Certain forest soils and soils with higher
initial organic matter contents may lose organic matter as a result of
altering the carbon-nitrogen ratio due to higher concentrations of nitrate
in waste water.

A variety of complex soil interactions through wetting of particular
clay types, changing organic matter content, and changing base saturation
has been reported in laboratory experiments (McGauhey and Krone, 1967).
In actual practice, however, a particular soil is much more susceptible
to changes in infiltration and percolation due to tillage practices
rather than subtle changes in wetting or soil chemistry. Soil surveys
for individual counties in each study area list the permeability in

inches per hour and frequently relate this permeability to soil textural classes. These data may be used to select soils with adequate moisture permeability rates if precautions are taken against physical clogging of soil pores with suspended organic material and a wetting-drying cycle which prevents formation of anaerobic conditions.

Spray irrigation disposal systems would require soils with a sustained percolation capacity in excess of 0.5 inch per hour. This percolation should allow passage of waste water at a quarter of an inch per hour and a safety factor for passage of natural rainfall at a quarter of an inch per hour. A successful overland runoff disposal system requires an almost impervious soil, (less than .2 of an inch per day). Rapid infiltration ponds require soils with a sustained percolation rate of 1-1/2 to 2 inches per hour and a sufficient vertical or lateral flow of several hundred feet.

Soil Surveys are published by the Soil Conservation Service, U.S. Department of Agriculture and available for most counties in the United States. These soil surveys list the major soil types usually in groupings of topographic situation such as soils of mountainous regions, soils of foothills or soils of alluvial plains. Soil types are often classed by infiltration, percolation and soil textural classes. These data may be used with confidence to select the proper soil type for a given disposal system.

Planners will generally find that design of spray irrigation disposal systems is the simplest means of waste water disposal with the highest potential for widespread success. The substantial land area requirement (10 square mile per 500,000 population) suggests that any contiguous disposal operation would cross many soil types. A flexible spray irrigation system could be designed to apply waste water at loading rates compatible with the soil texture and depth of each particular soil type. Good information currently exists on the proper irrigation management of many major soil types.

The overland runoff system is limited in application because extensive areas of relatively impermeable soils do not exist. The renovation ability of particular impervious soils is also not as predictable as the relationships of spray irrigation. Given a particular soil (for spray irrigation) with a given depth, we can predict with confidence the degree of renovation. The same is not true, however, with overland runoff or rapid infiltration ponds.

Of the three methods rapid infiltration ponds would probably be most difficult to operate and require the most precautions and checking. Rapid infiltration ponds are dependent on a greatly increased length of soil column to achieve renovation accomplished by spray irrigation in a relatively short soil column or overland flow on the soil surface. Coarse textured sands and gravels are required to achieve water flow rates of 1 to 3 feet per day. These soils have greatly reduced reactive surface areas and suggest a finite life of the column for renovation.

Another difficulty with rapid infiltration ponds is that very frequently we have much less knowledge of the chemical composition and physical structure of subsurface materials as compared with surface soils. By their very nature soils, which would lend themselves to rapid infiltration disposal methods, are very heterogeneous.

Water has high value in California for stimulation of crop production. Both spray irrigation and overland runoff (though not restricted to impermeable soils) are acceptable methods of irrigating crops. Spray irrigation might be used where direct contact of waste water with the crop is not objectionable. Overland runoff could be used where direct contact of waste water with the fruit or crop is not desired. Many soils in the California study area are sufficiently deep to expect complete renovation of waste water before contact with a groundwater table. Both methods of irrigation are in common practice and it would appear that utilizing waste water instead of usual irrigation water would offer no problems.

- 45 -

Soils recommended for spray irrigation disposal in the California area
are classed as soils of the foothills occurring along the eastern slopes
of the coast range and foothills of the Sierras. The Sehorn soils are
pale brown to brown, generally well drained, moderately deep, formed on
unaltered conglomerate, sandstones and shales of the Cretaceous period.
They generally have surface soil textures of a silty clay to clay loam
and a subsoil of silty clay. Contra Costa soils are brown to reddish
brown formed from sandstone and shales but usually of higher iron content.
These two soils are used chiefly as pasture and dry range in the rolling,
hilly areas of the Sacramento Valley.

Soils of the Nacimiento-Altamont-Shedd association also appear well
suited for waste water disposal by spray irrigation. These soils are
formed from calcareous sediments, are moderately deep to very deep and
usually well drained. Most of the area in this soil is used for dry
farming or range for livestock. Grain yields are high and the soil would
benefit greatly from irrigation water.

Few soil types exist in the California area that are sufficiently imper-
meable for application of overland runoff disposal methods. Most exten-
sive areas would require modification of the surface landscape by grading
to eliminate microdepressions, thus many soils might have their structure
sufficiently disturbed to become relatively impermeable. The Myers-Hillgate
association are soils with very low permeability. Grading and surface
compaction would be necessary for a successful overland runoff operation.
Currently, they are used largely for grazing or dry farming of barley.
With irrigation they are productive for sugar beets and corn.

Soils formed on recent alluvial lands and flood plains might be suitable
for rapid infiltration ponds. Undoubtably the underlayment for the ponds
would have to be artificially created and numerous test holes exploring
underlying material drilled. A series of bulletins is available from the
California Department of Water Resources which report groundwater depths
and quality on a county basis. These should be consulted for the design
of any disposal system but in particular the rapid infiltration ponds.

Rapid infiltration ponds appear the least desirable of the three methods of land disposal for the California area. Soils which will maintain rapid infiltration rates under ponding must have large soil pores found only in very coarse textured soils. Under these soil conditions retention of waste water in the soil columns would not be sufficiently long to achieve a high degree of renovation, thus, substantial amounts of dissolved materials could be contributed directly to the water table. As mentioned, movement of groundwater in a complex aquifer system then becomes very important.

The general considerations for the utilization of waste water as a substitute for irrigation apply in the Lake States in the same manner as California. The special constraint related to return of water to the Great Lakes Basin makes consideration of rapid infiltration ponds and overland runoff particularly appropriate. Rapid infiltration ponds could be utilized to a relatively shallow water table (5-7 feet). Water would then be removed from the water table at a rate equal to application to return diverted water to the Great Lakes Basin. Under these conditions sandy soils with high percolation rates might be utilized. Each soil would have finite capacity particularly for the renovation of nitrate. This might require frequent changes in the site of the infiltration pond or waste water loadings at less than the designed rate.

Compacted glacial soils on sloping sites could be used for overland runoff on soils with very low permeability. The Paris, Texas results suggest a lesser degree of renovation would be achieved using overland runoff on relatively impermeable soils. Terraces could be constructed for ease of accumulation of overland runoff and return to the Great Lakes Basin.

In the Lake States soils of the Zurich-Grays-Wauconda association and the Miami-Montmorenci association are well-suited for spray irrigation disposal. Both soils are on redeposited material usually having glacial origin. These associations occur in the Chicago area. In the eastern area (Ohio) soils of the Canfield-Wooster association are also well-suited for spray irrigation. Extensive areas of these soil types exist as rolling uplands

- 47 -

above any flood plain. Occasionally, slowly permeable pockets do exist but these are usually exceptions. Equivalent soils for infiltration ponds and overland runoff cannot be recommended, however, the same comments apply as offered for the California study area.

Soils in both the Lake States and New England area are frequently very *heterogeneous* as both underlying material and the surface have been extensively reworked by glaciation. In general soils in the Lake States have more acceptable soil textures (finer textured) than soils in New England.

The reduced evapotranspiration losses (20-30 inches) in the New England study area suggest a declining benefit from the economic utilization of waste water for irrigation. Evaporation exceeds precipitation from May through August with the greatest deficit in June. New England conditions suggest the greatest benefit from waste water disposal would be in augmenting streamflow with water of acceptable quality. Spray irrigation on sloping soils such as the Hermon, Paxton and Gloucester should be very suitable for spray irrigation disposal of waste water. These soils are all relatively deep and well drained, of somewhat coarser texture than might be desired. They are associated with a variety of current land uses which include forested dairying, poultry and in some cases orchards. Compacted glacial tills of *impermeable nature* do not occur over extensive areas. This would suggest that the overland runoff method of disposal may not be appropriate in the New England area. Many of the soil types have gravelly terraces which should be appropriate for rapid infiltration ponds. As with both the California and Lake States studies, however, *extensive reconnaissance* of the particular site would be necessary to establish subsurface flow patterns in gravelly materials as well as potential contamination of the groundwater.

CHAPTER VI - MICROORGANISM REMOVAL FROM WASTE WATER

The removal of microorganisms from waste water as it contacts the soil is
an important consideration in land disposal of domestic waste. How free
the percolated groundwater or overland runoff is of human pathogens
depends upon several factors. Of course the most important factor that
determines the residual number of organisms in an activated sludge effluent
is the efficiency and continuity of disinfection. However, considering
the greater resistance of viruses to routine disinfection procedure and the
possibility of chlorination failure and disposal of wastes not disinfected
increases the importance of the capacity of the soil itself to remove organ-
isms. Once removed the longevity of pathogens in the soil becomes of
interest. Potential for local contamination by air transport should also
be considered.

A. Efficiency of Soil for Filtering Viruses, Bacteria, and Fungi

A considerable amount of information from field observation indicates that
bacteria and viruses are efficiently removed from waste water as it perco-
lates through the soil. Viruses are probably transported to greater depths
in some instances than bacteria because of their smaller size (Drewry and
Eliassen, 1968); however, virus transport is also considered minimal and
some believe at no greater rate than bacteria (McGauhey, 1968). "Gravity
and times are against bacterial contamination of an outcropping ground water"
and unless fissures or dissolution channels are present for organism trans-
port, percolation through even the coarsest soil will remove bacteria and
viruses within a few to several feet (Ewing and Dick, 1970, McGauhey,
1968).

Experimental test plots at Whittier and Azusa, California showed a high
removal efficiency in even the coarsest soil. Coliform concentrations
in waste water following secondary treatment were decreased from 110-
120,000/100 ml to undetectable levels and 60/100 ml, respectively, at a
depth of seven feet.

McGauhey and Krone (1967) cite these results from the studies of A. M. Rawn and his associates. These authors review experimental work in infiltration ponds at Lodi, California that also showed nearly complete removal of coliform organisms within four to seven feet of a fine sandy loam soil. Concentrations in the effluent of 2×10^8/100 ml decreased to only 100s/ ml at 2 feet (Romero, 1970). However, this removal was nearly independent of the initial concentration in the waste water. Also cited were tests in dry soil around privies, where coliforms were not detected below depths of five feet where the water table was only 10-12 feet below the surface. Coliforms were removed less efficiently in coarse wet sand (0.15 mm diameter) but were still removed within 10 to 12 feet (Romero, 1970).

Although several feet of soil appear necessary for near complete removal of bacteria, most of the organisms are removed in a shallow film of soil at the surface. McGauhey and Krone (1967) cite the Lodi, California work (Orlob and Butler, 1955) to indicate the effectiveness of the uppermost 1 cm of soil in removing organisms. As much as 92-97% removal has been observed to occur in this upper 1 cm layer (Marculeseu and Drucan, 1962). These results are particularly significant in considering overland runoff as a treatment method in areas with soil of high clay content and low permeability. In such situations percolation may occur in the first few centimeters only as reported from the Campbell Soup Co. treatment site at Paris, Texas (Law, et al., 1969). Percolation may be minimal but since waste water flows continually over and through a few hundred feet of this "effective top cm" removal efficiency with overland runoff should be almost as effective as vertical seepage. This may be particularly true if other mechanisms besides physical filtration are considered, e.g., biological competition.

Removal efficiency (% removed per distance travelled) from waste water disposed in rapid infiltration ponds will probably be less than with either of the other two procedures of disposal. However, distance travelled and detention time will be greater, so ultimate removal may be as great as with spray irrigation. A couple sources of information exist that give an indication of organism removal under high rates of infiltration. At Santee, California, cited again in the review by McGauhey and Krone (1967), treated waste water was percolated through 1500-foot

- 50 -

channels composed of sand and gravel. The flow rate through these channels was about 100 feet/day. This, of course, represents a very rapid infiltration rate through a porous medium. Most of the bacteria in the effluent, and all viruses that were experimentally added, were removed in the first 200 feet of channel (Bureau of Sanitary Engineering, 1965, Merrell et al., 1965). At the Flushing Meadows Project near Phoenix, Arizona, waste water is added to infiltration basins consisting of three feet of fine loamy sand underlaid by a succession of coarse sand and gravel layers to a depth of 250 feet and clay deposits. This upper layer contains only 2% clay. The infiltration rate was 330 feet of waste water per year or about one foot per day. About 30 feet from the point of infiltration (8 feet vertically and 22 feet horizontally) total coliforms had decreased to about 200/100 ml during 2-3 week inundation periods to 5/100 ml during 2-3 day periods, a removal of greater than 99.9%. Fecal coliforms in this water were usually near zero. Underground detention time of about one month was considered adequate for complete removal of all coliforms (Bouwer, 1970).

The groundwater flow and hence detention time of the treated waste water in the ground is of considerable importance to reuse of the reclaimed water for drinking. Organisms are not removed as rapidly (percent removal per distance) from horizontally flowing groundwater as they are vertically in unsaturated soil. Organisms reaching the groundwater may travel more than 25 feet horizontally with a groundwater flow of 25 feet per day (Romero, 1970). Coliform organisms as well as enterococci travelled no farther than 100 feet from an aquifer recharge well following addition of varying amounts of primary sewage (McGauhey and Krone, 1954). Thus, travel distances of 150-200 feet should result in adequate organism removal from waste water by rapid infiltration. Detention time of several days over this distance seems advisable before reuse is possible.

The removal efficiency of microorganisms from waste water applied to soil would be expected to be great by any of the three methods; spray irrigation, overland runoff or rapid infiltration. Probably the least efficient removal would occur from overland runoff, although even here filtering capacity of particulates by the plant debris on the surface is fairly efficient. Rapid infiltration ponds would probably attain an

- 51 -

ultimate removal capacity similar to that of spray irrigation, but as mentioned earlier, it will require a greater distance and detention time.

The areas considered for spray irrigation treatment of secondary waste water are of varying soil type. The lower great lakes areas of Southeast Michigan and Northern Ohio are composed mostly of glacial till and sub-surface glacial moraines--heterogenous material consisting of fine to coarse texture. The permeability of such material is probably moderate compared to the other areas; San Francisco Bay-San Joaquin Delta area and the Merrimack River Basin. The areas in the Merrimack are considered to have very permeable sandy soil, with the San Francisco area containing generally more clay and are thus less permeable. Of course, much hetero-geneity exists in each area. From the standpoint of organism removal, all these soil types will probably perform adequately. Factors that will limit application rates of waste water on different types of soils and by different methods will probably not be microorganism removal.

B. Residual Pathogens

Human pathogens of most interest here that can be transported with feces in sewage are mostly included in Shigella, several of which cause bacillary dysentery, and Salmonella, which includes members that cause typhoid fever, paratyphoid fever and dysentery. Although many (150) viruses have been isolated from sewage (Bloom et al., 1959) only two have been known to be epidemiologically significant. These are possibly the causative agents for poliomyelitis, but definitely the causes of infectious hepatitis (Williams, 1971; Van Donsel, 1970). E. coli is of intestinal origin and occurs in large numbers, but usually is not considered a pathogen; its presence is used to indicate the presence of pathogens. Other members of the coliform group such as Aerobacter aerogenes is a normal complement in the soil and responds to the total coliform preferred test. For that reason fecal coliforms are often determined. These include organisms that will grow at human body temperature and thus indicate fecal contamination, although not necessarily of human origin. "Regrowth" of those organisms that normally reside in the intestine of warm-blooded animals should not occur in nature since they are not adapted to that environment (McGauhey and Krone, 1967), but regrowth can occur in members of the indicator group that normally occur in nature.

When organisms in waste water enter the soil, they are faced with competition for space, food supply and antibiotic materials from other microorganisms and also predation by larger soil organisms. Competition is apt to be greatest in this shallow surface layer since oxygen is more abundant and rates of decomposition are greater, which would lead to greater food supplies for larger organisms in that layer. Thus, human pathogens not adapted to the rigors of such an existence may not be expected to survive long. This has been demonstrated to some extent in that E. coli survival was considerably greater in sterile soil than in non-sterile soil (Rufolfs, et al., 1950). McGauhey and Krone (1967) cite work that shows low survival of organisms from human intestinal origin as a result of competition with normal soil microflora. The extent to which biological competition and simply physical filtration contribute to microorganism removal from waste water in each case is not known. However, in some instances survival in soil of organisms from intestinal origin has been relatively great. Since removal efficiency seems to be high in nearly all soil types examined, and no great variation occurs with seasons and temperature, one might conclude that physical filtration is the principal removal mechanism. The persistence of these organisms once removed by the soil depends on biological controls regulated by environmental factors such as temperature.

The persistence of these organisms in the soil is then of considerable interest in the event that contact with people is possible. However, such direct contact is even less probable than contact by transport to groundwater and eventually to surface water. The concentrations of pathogens remaining in the soil would probably be low because as considered here, disinfection of the waste water is a prerequisite to land disposal. Disinfection may not actually be necessary in all cases of land disposal (McGauhey, 1968), but it is a conventional practice following activated sludge treatment. Thus, the concentration of organisms in waste water applied would be the tolerant residual following disinfection.

Chlorine is the standard disinfectant applied to waste water at present, although other disinfectants are being tested and considered, such as ozone. Of course with any toxicant there exists a differential tolerance within the population. Therefore, under routine practice of disinfection

- 53 -

one would not expect a complete kill of everything all of the time. In
early work Butterfield (1948) determined that a combined chlorine
residual of 1-2 ppm over a pH range of 6-10 and a contact time of 60
minutes was considered necessary to obtain a 100 percent kill of bacteria.
Although viruses are much smaller some are more difficult to kill than
bacteria (Mallman and Mack, 1961). Williams (1971) suggests that viruses
are killed under normal disinfection procedures with chlorine--1 ppm
residual with 30 minutes contact time. Chambers (1971) reviews findings
to the contrary.

Because pathogens, and particularly viruses, are difficult to measure,
coliform bacteria (total and fecal) are used as indicators of disinfection
effectiveness on a routine basis. Although the effectiveness of pathogen
removal, particularly viruses, and the validity of the coliform index is
a controversial subject, it appears that normal disinfection practices are
reasonably effective at removal of pathogens (including viruses). Under
routine operation Chambers (1971) reported results of coliform survival from
45 treatment plants. Most (93%) of the plants showed that less than 5000/
100 ml survived disinfection and over one half (56%) showed that less
than 500/100 ml survived. This represents at least 99.9% removal in most
plants.

The longevity of such an extremely small percent residual bacterial
concentration in the soil will vary depending upon environmental conditions.
Temperature, organic matter content and whether or not the soil is aerobic
are the more important variables. The Salmonella typhoid bacillus has
been observed to survive in soil from one half to a full year (Rudolfs
et al., 1950), but McGauhey and Krone (1967) cite work that shows maximum
survival of about one month. Survival increased with increased organic
content in the soil--survival was decreased to less than one week when
sand was used.

Coliforms survive longer on the average than typhoid or tuberculin bacilli
(Rudolfs et al., 1950). E. coli and A. aerogenes have been reported to
survive up to four years in soil (Mallman and Mack, 1961). Moisture was
found to be significant--survival in wet soil was double that in dry
soil (Rudolfs et al., 1950). As stated previously, biological competition
and predation are important determinants of enteric organism longevity in
the soil and consequently surival can be expected to be greatest when

- 54 -

normal biological activity is least, i.e. low temperataure, anaerobiosis,
etc. Thus, findings that _Salmonella_ survive two years in frozen soil, but
a maximum of one year in unfrozen soil, and that _E. coli_ survive longer
in sterile than in non-sterile soil are not surprising (Rudolfs et al.,
1950).

Although longevity of pathogens could be significant during the beginning
and ending of the spray period in cold climates, survival would not
usually be expected to exceed one month if biological activity is relatively
great. If death and decomposition rates and loading rates of pathogens
are constant, significant accumulation of pathogens would also not be
expected even though longevity can be a month or more.

Air dissemination of surviving pathogens from waste water irrigation
sites should be considered as a possible hazard even though the likelihood
of disease spreading from such sites would at first seem remote. Spray
irrigation would probably expose a small percentage of the particles,
including bacteria, in the waste water to action of the wind. This
presumably minimal opportunity for wind removal of particles from sprayed
droplets together with the very low probable concentration of pathogens
in the waste water suggests that the hazard of airborne transmission of
disease from spray fields (where waste water is disinfected) would be
slight. Measurement of air dissemination of enteric organisms from sites
of land disposal of waste water are limited, but Sepp (1965) notes that
in one instance coliforms were detected as far as 200 feet from the point
of application. However, extra safe precautions would seem in order and
residential areas should be kept at a considerably greater distance from the
spray fields than would be suggested from considerations of groundwater
contamination and reuse alone. Even though dilution is great and survival
poor, transport of disease-causing microorganisms from concentrated point
sources has been observed to be great and should be carefully studied
on representative land disposal sites.

CHAPTER VII - BOD REMOVAL AND WASTE EFFECTS ON SOIL ORGANISMS

The activity of waste and soil microorganisms in response to waste water addition to the soil greatly affects the quality changes that occur in that water as it percolates through or over the soil. Of concern is the decomposition or removal rate of BOD from the waste water. The capacity of the soil as a physico-biological remover of BOD is great. This capacity can be exceeded, however, by adding a greater mass of BOD than can be decomposed within the time that the water is detained in the soil or by applying more waste water than the soil will absorb (hydraulic overloading). In both cases anaerobic conditions may result which would greatly reduce the decomposition rate and increase the possibility of clogging. Clogging can result from various processes, but the effect of biological activity seems to be the most important (McGauhey and Krone, 1967). Although clogging would not be expected on well-managed disposal sites at an application rate of two inches per week, it seems of value to consider the conditions that would cause such a problem to better understand the capacity of soil for waste water treatment.

The BOD removal efficiency of soil can be affected by the amount of vegetation cover and the infiltration capacity. Anything that adds surface area at the soil air interface will increase biological decomposition capacity whether it be litter or living plants. At very high rates of waste water infiltration residence time of the dissolved or particulate BOD may not be great enough for complete biological decomposition to occur and a sizable fraction of undecomposed BOD may reach the ground water. However, net BOD removal efficiency has been observed to be quite high even in the coarsest soils and highest infiltration rates studied, even though the removal efficiency per unit soil column decreased under such conditions.

Of additional interest is the effect of increased loading of inorganic and organic nutrition and toxins on the activity and abundance of indigenous microflora. In general, all forms of heterotrophs and autotrophs alike might be expected to show an increase with increased available nutritive material. Only when such material begins to reach the exhaustion stage will certain forms more competitive at low concentrations begin to be favored. A shift

in the kinds of organisms most abundant and their activity may be more likely to occur then at low or intermittent BOD loading than at high and constant loading. Of course, organic and inorganic toxicants at even low concentrations may cause shifts in dominant organisms and/or a total decrease in abundance and activity of all forms if the amount and effectiveness is great enough. This is largely speculation and based on experience in aquatic environments but, along with available information, will be discussed below.

Commonly, microbial populations are treated as consisting solely of bacteria and algae when, in reality, under natural plant ecosystems fungi occur quite prominently. In most soils bacteria are found in numerical preponderance; however, fungi, due to size superiority, are thought to account for the largest part of the microbial protoplasm. It has been estimated (Clark) that the biomass of fungi in agricultural soils occur at the rate of more than a ton per acre. Since living protoplasm of soil systems is the machinery for carrying out living processes, the chemical conversions necessary for abatement of pollution due to sewage disposal, growth and reproduction of fungi are therefore important components of any soil ecosystem. Of the important soil microbial reactions necessary in the functioning of a waste water soil disposal system, fungi are capable of accomplishing a wide range of chemical reactions. In fact, the fungi have been, in certain aspects, found to be most efficient in many chemical conversions under aerobic soil conditions. However, fungal reactions are critically affected by environmental factors such as pH, temperature, oxygen tension, carbon/nitrogen ratio, carbon/sulfur ratio, etc.

A. Mechanism of BOD Removal

Biochemical oxygen demand (BOD) is removed from waste water in the soil mantle by a combination of physical and biological processes. The soil acts as an effective filter in removing particulate matter--most of this removal occurring in the upper 5-6 inches in the profile (McGauhey and Krone, 1967). Physical removal can occur also by adsorption of dissolved organic compounds. Law et al (1969) have shown that this physical

- 57 -

removal can be as much as 30-40% of BOD, COD (chemical oxygen demand) and TOC (total organic carbon).

The particulate organic matter that is filtered by the soil as well as that dissolved in the percolating water will be partially degraded by microorganisms. Soils contain a large complement of heterotrophic micro-organisms allowing the total system the ability to utilize and degrade a broad array of organic compounds under a variety of environmental conditions. Most materials will eventually degrade--the time required, however, may range from minutes in the case of glucose, to hundreds of years in the case of complex aggregate of compounds called soil humus (Tamm and Ostland, 1960). Under aerobic conditions the bulk of the degraded mass will evolve from the system as CO_2. If the active surface layer of the soil is anaerobic, degradation will stop short of complete conversion to CO_2 and many reduced organic compounds will result from partial degradation. Under both conditions, mineralized inorganic materials will result from the process and either be taken up by plants, held in the soil, or more likely, transferred through the soil mantle with the water flow. The degree to which these will be removed from the soil and enter the groundwater will determine, in part, the environmental effect of the waste water. A portion of the degraded organic matter will be reconstituted into biological cells, sometimes referred to as sludge, which can contribute to food for larger animals such as insects or annelids (worms), a substrate for further microbial decomposition or clogging problems in the soil mantle.

Soils bearing different higher plant populations, i.e., agricultural crops, forest, or grasses and shrubs, generally exhibit characteristic microbial populations. Each microbial population is capable of handling a wide range of chemical reactions typical of the chemistry of the given soil of the ecosystem. As pointed out by Clark's (1967a) priming concept, such microbial populations will respond to gradual changes in soil environment, i.e., pH, temperature, and nutrition, and maintain a high degree of activity if the factors of the environment do not change drastically. Changes in the chemical characteristics or quantities of input into a soil within relatively small periods of time can lead to an

- 58 -

overloading of the soil system. Such overloading of the soil system with
a liquid effluent simply may limit the available oxygen (DO) required
for the functioning system. These actions result in an extended lag
phase or may totally destroy the biological system previously developed
to the desired level of activity.

The nature and percentage of organic matter in secondary waste that is
included in its BOD is of interest because it is the relatively refractory
materials that will be most apt to resist removal in the soil mantle and
may reach the groundwater. Although little is known of the kinds and
concentrations of organic compounds in secondary waste, McGauhey and
Krone (1967) reviewed the limited amount of knowledge on the subject.
These authors cite work that shows only about 30 percent of the organic
carbon in secondary treatment effluent could be assigned to chemical
groups (Painter, 1961; Bunch et al., 1961). Whereas 50 percent of the
soluble organics in settled sewage are carbohydrates only 10 percent
could be ascribed to this category following secondary treatment. Most
compounds after that stage were considered to be soluble acids.

From the above data, then, the immediate carbonaceous BOD in secondary
effluent is probably a small fraction of the total dissolved organic
matter.

McGauhey and Krone (1967) conclude from results of several workers that
simple sugars, starches, hemicelluloses, celluloses and proteins decompose
relatively rapidly in soil, whereas such groups as lignins, waxes, tannins,
cutins and fats are more resistant to decomposition. If anaerobic condi-
tions are created by too great a hydraulic or BOD loading for the soil
capacity, then these more resistant compounds are apt to accumulate.

B. Effect of BOD Loading

Considering the model secondary effluent with a BOD concentration of
25 mg/l and an application rate of two inches per week results in a
hypothetical BOD loading rate of 1.64 pounds/acre/day (11.5 lbs/a/wk).
This is a relatively small loading rate and would not be expected to
result in appreciable accumulation of undecomposed organic matter in

- 59 -

the soil mantle. Since soils have remained aerobic under loading rates of hundreds of pounds per acre per day with percentage removals in the upper 90's, it would appear that treatment of the model waste water would be limited more by hydraulic capacity of the soil than by the ability of its microflora to remove the BOD. Since disposal sites will be selected to adequately accommodate hydraulic loading of two inches per week, BOD removal from spray irrigation and rapid infiltration is expected to be highly efficient. Efficiency may be less with overland runoff. The following literature review should tend to support this contention.

Industrial waste water containing 1,150 mg/l BOD_5 was sprayed on Vilas sand with fairly high permeability and well covered with Reed Canary grass and humus (Philipp, 1971). BOD removal was reported greater than 99% with a BOD loading of 138 lb/acre/day (966 lb/a/week). Based on preliminary experiments, it could be concluded that effective operation of the system was hydraulic loading. At three inches per day the soil remained too wet and the Canary grass, very important to the soil's percolation capacity, was killed. This work indicated that the treatability of a large volume waste with low BOD may be limited by percolation capacity of the soil, while a small volume waste with high BOD is more apt to be limited by the oxidative capacity of the microorganisms and sorptive capacity of litter on the soil surface.

Waste water from the Union Bag-Camp Paper Co., in Franklin, Va., was sprayed during eight months of the year onto Norfolk-type sand (infiltration rate 4-6 in/day) at the rate of 3.6 in/week. This represents a loading rate of 181 lb/acre/week, at a BOD concentration of 219 mg/l. Although from 3,889 to over 6,340 lb of BOD were applied per acre during this period, no change in the organic matter in the soil was observed (Crawford, 1968).

Thomas and Bendixen (1969) experimented with BOD loading rates, soil types and temperature in the treatment efficiency of septic tank and trickling filter effluent. The trickling filter effluent had a BOD of about 46 mg/l, 67-68% of which was removed by percolation through silica

sand at BOD loading rates of 560 and 140 lbs per acre per week. Although the percent removal was similar at the two loading rates, 10 times as much residual BOD escaped through the system at the higher loading. At 140 lbs BOD/acre/week 89% removal occurred in a silt-loam soil compared to the 68% in silica sand. The surprising result in experiments using septic tank effluent (BOD about 93 mg/l) was that the temperature range (28° C, 18-35° C, -18-35° C) or a range in dosing period from 82-425 days did not affect the BOD removal efficiency (~77%). Jones and Taylor (1965) also had found that BOD removal from septic tank effluent (BOD, 41 mg/l) was consistently about 80% over a range of three sizes of sand in experimental columns with infiltration rates of 32, 22 and 10 inches/hour. Gravel was considerably less efficient at removing particulate matter.

Elazar et al. (1971) cite several examples of waste water treatment by application to soil and in all cases BOD loading is relatively great and removal efficiency high. The results at these sites may be summarized in Table VII-1. The consistently high removal efficiency of BOD from waste water applied to soils by spray irrigation is evident for a variety of soil types and application rates. Only in the case of very coarse soil with exceptionally high infiltration rates would the physico-biological mechanism fail to remove most of the BOD within a fairly shallow surface layer of soil. However, rapid infiltration is ultimately very efficient at BOD removal. At the Flushing Meadows Project at Phoenix (Bouwer, 1970), BOD in secondary plant effluent is reduced from 15 to 0.3 mg/l (98%) over a combined vertical and horizontal distance of about 30 feet. The application rate here ranges from 1 to 4 feet per day. The soil type there is represented by 3 feet of sandy loam over 250 feet of sand and gravel.

Overland runoff is expected to provide a less efficient removal of BOD than the other two methods of disposal. Waste water from the Campbell Soup Co. plant at Paris, Texas, is applied to rather impermeable soil, and treatment is accomplished by overland runoff. In a study of the system conducted by Law et al. (1969) 133 inches of waste water was

Table VII-1

SUMMARY OF DATA FROM SITES OF WASTE WATER TREATMENT BY SOIL APPLICATION

(Elazar et al, 1971)

Activity	BOD mg/l	Application Rate in/wk	BOD Loading lbs/ac/wk	Soil Type	Removal Efficiency
Seabrook Farms - Food Process	1000*	14	3220	loamy-sand	98%
Riegal Paper Co. - Industrial	600	3.85	531	sandy	95%
Nat'l Fruit Products - Food Products	2600	8.75	5232.5	sandy-loam	99%
Bearmore & Co. - Industrial	600	8.4	1159	glacial till	95%

*Estimated considering usual high BOD of food process waste

applied to four experimental watersheds over a 43-week period. The average BOD in the waste water was 572 mg/l, the application rate was 3.08 inches per week with a resulting BOD loading of 405 lbs/acre/week (57.9 lbs./acre/day). Very high removal efficiencies (>98%) were observed in the system even though percolation into the soil below the first inch or so was slight (percolation rates of 0.12-0.17 in/day). The high efficiency at this site is partly a function of the high BOD (572 mg/l) applied. Probably because of less filtering capacity by this method, a fairly high residual BOD remained (mean 9 mg/l).

A similar residual may also be expected with the model waste water even though initial BOD is low.

Clogging is not expected to occur from applying secondary waste water to soil at two inches per week, but a discussion of the mechanisms and consequences is included to understand the range in treatment capacity of soils. Although clogging can result from biochemical reactions, loading of inorganic and organic materials (both particulate and dissolved), excessive hydraulic loading and geometry of the soil surface and profile, the most significant cause is the activity of microorganisms (McGauhey and Krone, 1967). The conversion of dissolved organic matter into biological sludge (cells) as well as the failure of microorganisms to decompose filtered organic particulate matter can similarly clog soils and, thus, greatly restrict infiltration rate.

Clogging usually occurs in the top few inches of soil (Jones and Taylor, 1965; and Laak, 1970) and is more a function of the organic mat which is largely independent of the coarseness of the soil (McGauhey and Krone, 1967). The perpetuation of anaerobic conditions in the soil surface layer will lead to clogging. Anaerobic conditions result in a slowed rate of biological activity and, thus, a tendency for sludge accumulation, production of ferrous sulfide and/or accumulation of polysaccarides. The anaerobic conditions could be brought about by either too high a BOD or hydraulic loading, but usually it is hydraulic overloading which leads to inability of the soil to accept the water application rate and ponding results. The "off-on" application procedure allows for drying out and

- 63 -

re-establishment of aerobic conditions in the soil. If "ponding"
persists clogging problems will increase.

In further results from the experimental column described earlier, Jones
and Taylor (1965) found that sand and gravel clogged 3 to 10 times faster
under anaerobic conditions than if aerobic. Continuous ponding of
applied septic tank effluent under anaerobic conditions decreased
conductivity through the columns. Laak (1970) also observed that domestic
waste water (untreated) rapidly clogged soil in experimental columns.
Clogging was attributed to bacterial cell (sludge) formation in response
to BOD consumption. Bacterial cells comprised 90% of the material that
completely clogged the soil surface layer after 180 days. BOD + TSS
(total suspended solids) was inversely related to in-service time of the
soil column; a 50% reduction in BOD + TSS content allowed 25-40% increase
in the hydraulic loading without changing the in-service time.

That soil clogging is a result of biochemical processes by organisms
within the soil and not a result of filling soil spaces with organisms
added with the waste water was demonstrated by Allison (1947). He found
that continuous inundation of the soil resulted in clogging and was
therefore a result of internal processes and obviously not caused by
added organisms. A typically biologically clogged soil may appear as a
heavy overgrowth of black biological slimes as illustrated by McGauhey
and Krone (1967). This slime layer is not all bacterial cells but, to a
large extent, comprised of ferrous sulfide. However, FeS was found to
penetrate to considerably greater depths than the biological mat
(Winneberger et al., 1960). Since the area of clogging is generally
considered to be the surface few inches, it was suggested that the bio-
logical mat was relatively more important in clogging than anaerobically
formed ferrous sulfide.

Polysaccharides and poluromides are also heav   :mpli..ted as being
responsible to a considerable extent for soil clogging. The accumulation
of these compounds in the soil has been inversely related to the infiltra-
tion capacity. These compounds are produced by microorganisms utilizing

readily decomposable organic matter added to the soil. The compounds
accumulate under anaerobic conditions but are rapidly broken down by
aerobic organisms when the soil is aerated (Mitchell and Nevo, 1963;
Mitchell et al., 1965; and Avnimelech and Nevo, 1964). According to
McGauhey and Krone (1967), the process of ferrous sulfide and poly-
saccharide accumulation helps to explain why continual loading of soil
will lead to clogging, but by employing resting stages for the soil to
dry out, the clogging problem will disappear. This is apparently because
the anaerobic condition brought on by the continual loading will become
aerobic upon drying (Thomas and Law, 1968). Although temperature was
not found to be particularly critical to the success of polysaccharide
consuming organisms, pH was very critical and must be maintained between
7 and 9.

The necessity of holding moisture content down to attain maximum decompo-
sition and minimum humus buildup has been shown by Shvachkin and cited
by McGauhey and Krone (1967). At 60% of moisture capacity 42% of the
litter was decomposed, at 100% moisture only 17% was decomposed and
only 8% was decomposed when the two moisture conditions were reversed
every ten days. Decomposition rate was greatly dependent upon pH also,
with neutrality affording the most desirable conditions. Rate dropped
off as acidity increased (references cited by McGauhey and Krone, 1967).

From the preceding evidence, it becomes clear that conditions of water
ponding and continual addition of waste water must be avoided. Even the
loamy, glacial till overlying fine sandy loam in the Merrimack Watershed,
which is moderately permeable, would require periods of drying out to
restore aerobic conditions and prevent soil clogging from becoming a
problem. Soils in the other areas are presumably less permeable than in
the Merrimack. Since clogging problems were apparently not observed at
the various sites that received exceptionally high BOD loading rates
(thousands of pounds per acre per week), it is unlikely that BOD loading
of secondary effluent as proposed (11.5 pounds per acre per week) will
contribute enough substrate to produce clogging problems provided that
aerobic conditions are usually maintained. As the evidence indicates,

- 65 -

hydraulic overloading by itself can produce anaerobic conditions and
clogging as a result of biological processes within the soil, regardless
of BOD or organisms added. The infiltration capacity of soils in the
three general areas is considered adequate to accept the hydraulic
loading of two inches per week, with frequent intervals of drying and
restoration of aerobic conditions. Therefore, clogging problems will
probably not occur under these conditions.

C. Effect of Nutrition, Toxins, and Loading

Many factors of the soil express effects on microbial reactions, heavy
metals, toxic organic and inorganic compounds, etc. However, if the soil
environment is maintained to provide a relatively high level of microbial
aerobic action, the normal concentrations of inhibitory substances ex-
hibited in typical secondary sewage treatment effluents has not been
found to induce detrimental effects to the living systems in soil (Sopper,
1971). However, shock loading of such inhibitory substances, especially
of an inorganic nature, should be avoided.

In aerobic soil environments fungi are often able to spread within a soil
system faster than bacteria. Therefore, positive concern must be expressed
for the effects on soil fungi responses to waste water input within a soil
system. The rate of waste water input must be governed by the damage po-
tential (soil microorganism tolerance to changes), of a given soil, other-
wise dissolved oxygen will become limiting to the point that carbon and
nitrogen utilization will be inhibited. During the processes of system
upset, imbalance can develop between biomass production and decomposition
rates that can be expressed as excessive loading.

Of special interest to loading effects are those of pH control and temper-
ature. It appears from experience (Cooke, 1970) with microbial actions
in soil, if input waste water pH is roughly neutral, biological systems
are not usually adversely affected. Alkaline pH changes are more detri-
mental to fungal action than acid changes.

Studies on soil microbial oxidative capacities showed small detectable influences by temperature changes. This was due primarily to the continued efficiency of the functioning biological systems. Basically, this was brought about by the increase in the fungal and bacterial populations with decreasing soil temperatures. Even though rates of physiological processes are slowed by decreasing temperatures, the increase in living biomass evidently maintains the total system efficiency. (Law, 1969)

Many organics reported (see Chapter IX-B) as constituents of waste water effluents are known to be biodegradable within certain soil systems. However, the exacting environmental conditions under which these reactions occur are not usually known. In reality, this knowledge is not absolutely necessary at this early stage of consideration, however, it would be highly desirable to develop a program of research to accumulate information on factors controlling the biodegradation within soil systems of organics found in waste water effluents.

Excluding carbon sources of energy, which are considered in BOD utilization, the items of oxygen and nitrogen in the fungal component of microbial nutrient cycles and phenol, as a model toxin, are reasonably well known and should be briefly discussed.

Oxygen: This item can be considered an expression of the microbial ecologists' factors of competition. Oxygen is an essential item in an aerobic microbial soil system functioning at the desired level to handle the waste water loading that given soil drainage characteristics will allow. If field moisture capacity is approached.for a given physical soil system, oxygen, dissolved and otherwise, becomes limiting forcing the biological system to become anaerobic, thus reducing the hydraulic loading capacity that can be managed by the biological systems. This then leads to a total loading problem. Available oxygen governs the rate of utilization of several important items that an efficiently operating soil biological system is capable of handling within a waste water disposal system, i.e., nitrogen utilization in its various forms, ABS compounds, carbohydrates and many other biodegradable chemical characteristics of waste water.

It is recommended that the measure of an efficiently functioning bio-
logical soil system for handling the disposal of any waste water can be
monitored by the evaluation of the dissolved oxygen of the soil system.
When this value is determined to be too low it would indicate loading
is rapidly approaching capacity from the physical, chemical and biological
system points of view.

Nitrogen: Like many other soil microorganisms fungi readily decompose,
as sources of nitrogen, proteins, amino acids, and other nitrogenous
compounds with the liberation of ammonium. A more complete picture of
the nitrogen turnover cycle as accomplished by microorganisms in general
is illustrated in Figure VII-1.

No single pattern of nitrogen assimilation can be described to apply to
all fungi. Carbon supply and availability of oxygen play major roles
in the assimilation processes. Nitrate reduction by fungi is believed
to be strictly assimilatory in nature whereas bacteria have been found to
reduce nitrates without assimilation and therefore each type of organism
will demonstrate a different soil loading response to nitrate additions
in a soil environment.

When nitrogenous compounds are added to the soil, numerous reactions
are triggered. The portions controlled by microbial metabolism contain
several transformations. First, the organic compounds are decomposed
and converted into ammonium and nitrate. In the environment nitrogen is
assimilated almost entirely in the inorganic state. (Alexander, 1961)
Simultaneously, there is also a limited amount of microbial utilization.

Many fungi readily decompose proteins, amino acids and other nitrogenous
compounds with the liberation of ammonia. The ammonia, in turn, is
oxidized by specialized bacteria (Table VII-2) to nitrate. This compound
is utilized, converted to gas or leached from the soil. In this respect
nitrogen is used in the soil. The fungi can readily obtain their energy
from carbohydrates, using inorganic salts as sources of nitrogen
(Cochrane, 1958). Actinomyces and heterotrophic bacteria prefer proteins,

- 68 -

Figure VII-1

Organic nitrogen

non-symbonic
fixation by the
Azotobacter group

Animals

Microbial decay produces
some organic nitrogen for
microbial cells, and releases
NH_3

Atmospheric N_2 $\frac{symbonic}{fixation}$ Plants
nitrogen

NO_3^- anaerobic* NH_3

1. This is utilized by the
   vegetation

2. Oxidized to nitrate

1. Lost by leaching

2. Plant nutrient

3. Reduced to ammonium

4. Converted to N_2, Atmosphere

* This process proceeds best under anaerobic conditions.

** The process from NH_3 to NO_3^- is a transfer by microorganisms; Nitrosomonas,
which oxidizes NH_3 to NO_2, and Nitrobacter, which finishes the oxidation to
NO_3.

peptones and some peptides as energy sources--particularly over poly-
saccharides. This difference, although there is much overlap, explains
the difference in behavior of these organisms towards various organic
compounds added to the soil.

Nitrogen in the nitrate form is easily lost from the soil in several
ways. Because of its solubility in the soil solution, nitrate is re-
moved by water. On a red pine forest soil, which received 2 inches of
waste water a week, water samples taken at four feet after three years
showed a decrease in nitrate concentration by 50% (Kardos, 1968). Some
of this nitrate is removed by the plant cover, some by denitrification.
Denitrification removes the soil nitrogen by returning it to the atmos-
phere; this is done both under aerobic and anaerobic conditions and is
important in clearing the amount of nitrate from the soil (Sopper, 1971,
Series 24). Because nitrite is rarely found, in nature, the nitrate
formers generally occur in the same environment as the ammonium oxidizers.

Forms Utilized: As a consequence of mineralization, ammonium and nitrate
accumulate and organic nitrogen disappears. Ammonium is typically asso-
ciated with a waste product overflow in microbial metabolism, the accumu-
lated ammonium representing the quality of substrate nitrogen in excess
of the microbial demand. Nitrification, however, is associate with the
energy-yielding reactions in the metabolism of autotrophic bacteria.

Limiting Factors of N Utilization: Reduction in the O_2 supply where it
is inadequate for microorganisms results in little ammonium oxidation and
under anaerobic conditions the reaction ceases. Because of this nutritional
dependency on oxygen, soil structure will affect the accumulation of
nitrate through its influence on aeration. Oxygen is an obligate require-
ment for many of the soil organisms (Cochran, 1958).

Because moisture affects the aeration regime of soil, the water status
has an important influence on the microbial flora. The optimum moisture
level varies considerably with different soils, but the best conditions
of optimum growth appear to be at one-half to two-thirds of the field-
holding capacity. (Sopper, 1971)

In respect to nitrification, temperature has a marked effect. Below 5°C and above 40°C the rate is very slow. The optimum again is very dependent but appears to be between 30-35°C (Cochrane, 1958).

Nitrogen has been demonstrated to play a vital role in BOD reduction processes. This is especially the case if the carbon source is of complex structure such as cellulose. Mineral nitrogen requirements are in critical demand until mature populations of microorganisms (fungi) develop. Thereafter, nitrogen is slowly released by autolysis of this population as the carbon substrate declines (Trive, 1961, from Burges).

Phenols: A wide range of normally biologically toxic organic compounds, such as quinones, ketones, phenols are known to be acted on by soil fungi to the extent of being detoxified within a soil system. In fact Clark (1967) goes so far as to state that, "if a compound is produced by living organisms, it can be acted on by a soil microorganism".

Specifically, two groups of soil fungi are known (Waksman, 1927 and Nei, 1971) to act on phenolic compounds normally functioning as biotoxicants. A group of the Basidiomycetous fungi known to induce white-rot type of wood decay are capable of oxidizing phenolic substances related to the structure of lignin. A member of this group (Fomes annosus, a soil inhabitant) has been demonstracted by Driver (1963) to withstand direct contact with high concentrations of chlorinated phenols and retain its ability to spread a root-rot disease within a forest.

Nei (1971) has recently demonstrated that a member of the red yeast fungi (Rodotorula glutinis) is capable of oxidation of phenol at maximum rates in the presence of suitable carbohydrate carbon (sugar) source under aerobic conditions. Cooke (1970) has demonstrated this red yeast group to be common in soil conditions associated with soil sewage systems.

CHAPTER VIII - EFFECTS OF INDIGENOUS PATHOGENS IN SOIL

CHAPTER VIII - EFFECTS OF INDIGENOUS PATHOGENS IN SOIL

A. Population Dynamics

Cooke (1970) and several other workers (Ajello, 1954, Grayston and Furcolow, 1953, and Davis, et al, 1970) have demonstrated the presence in soil of many pathogens of humans, wildlife and domestic animals and plants. These organisms are thought to usually be in a resting stage within the soil system. Favorable changes in the soil environment such as moisture, nutrients, temperature, pH, and other factors providing for initiation of growth and reproduction of the resting propagules of bacteria and fungi capable of functioning as pathogens would increase the potential for inducing disease within   susceptible host populations. Cooke (1970) has demonstrated for some fungal pathogens of humans and plants capabilities for existence and growth in soil environments associated with sewage effluents. Refer to Table VIII-1 for a list of the pathogens reported to date that have been found in soils associated with urban areas.

B. Potential Spread of Pathogens

Practically all of the organisms listed in Table VIII-1 are capable of being spread by wind currents and many are known to be spread by overland flow of water. To date, few problems have been reported resulting from the use of sewage effluent on agricultural crops. However, this practice is relatively new in this country and it has been predicted by Menzies (1954) that certain plant pathogens will increasingly cause problems as the practice of irrigation increases. This report was in relation to air-borne plant diseases. Garrett (1938) predicted, and further studies have shown, that high levels of moisture are favorable to plant pathogens dependent on water for dissemination of zoospores, such as species of Phycomycetes. In addition, many pathogens benefit from host susceptibility induced by high moisture.

Little is known of soil-borne viruses pathogenic on plants, however, Cadman (1963) reports that soil moisture regulates the abilities of these organisms to survive in agricultural soils. Most known plant disease, soil-borne

virus pathogens maintain infectivity under most soil conditions. One group, those transmitted by ectoparasitic vectors, lose infectivity on drying of soil.

Merrill's, et al (1969) extensive review of this area points out the critical need for research on the specific subject. In the case of plant diseases this group stresses the fact that much of the information available for evaluation of disease hazards consists of gross observations only on relatively short-term studies. Few detailed studies have been made on the biological portions of irrigation ecosystems. The limited research available on the effects of irrigation versus plant pathogens all points in the direction of increased problems.

Table VIII-1

Pathogenic fungi associated with urban soil systems (Cooke, 1970)

Human pathogens:

Allescheria boydii
Aspergillus fumigatus
Geotrichum candidum

Plant pathogens:

Alternaria tenuis
Aspergillus flavus
Aspergillus fumigatus
Aspergillus niger
Aspergillus terreus
Botrytis cinerea
Cephalosporium spp.
Chaetomium funiculosum
Chaetomium globosum
Cladosporium cladosporioides
Coniothyrium fuckelii
Curvularia lunata
Fusarium aquaeductuum
Fusarium moniliforme
Fusarium oxysporum
Fusarium roseum
Fusarium solani
Gliocladium catenulatum
Gliocladium roseum
Mucor - several species
Penicillium brevi-compactum
Penicillium chrysogenum
Penicillium cyclopium
Penicillium digitatum
Penicillium expansum
Penicillium italicum
Penicillium martensii
Penicillium oxalicum
Penicillium rugulosum
Rhizopus nigricans
Scopulariopsis brevicaulis
Stemphylium consortiale
Trichoderma viride

CHAPTER IX - CHEMICAL INTERACTIONS OF SOIL AND WASTE WATER

CHAPTER IX - CHEMICAL INTERACTIONS OF SOIL AND WASTE WATER

Adsorption, absorption and ion exchange reactions when combined with
dissolution, volatilization, precipitation, complex formation and redox
reactions together with microbial degradation and absorption, plant up-
take, and physical retention processes (filtering) provide mechanisms
by which soil solution concentration are controlled in the soil-water
system. In general, mechanisms of renovation of waste water in soils
are relatively the same for both inorganic and organic materials. The
fate of **waste** water materials in soils is determined by a large number
of processes, including adsorption on solid surfaces, plant and micro-
bial uptake, microbial degradation, volatilization, leaching, chemical
breakdown and precipitation. The relative importance of each of these
individual mechanisms in waste water renovation in soils will ultimately
be a function of type and concentration of the material and soil chemistry.

A. Mechanisms of Renovation

The following section briefly reviews factors and mechanisms that may
contribute to waste water renovation in land disposal systems. The
major transport agent of materials in soil is water. Materials are trans-
located through the soil profile into groundwater and are carried in surface runoff.

Soil factors that determine effectiveness of waste water renovation are:
Clay minerals, organic matter, soil permeability, soil pH and depth to
groundwater. The soil's ability to renovate waste water depends, in
addition to soil characteristics, on design, type, configuration and loading
of the application system and maintenance of the land disposal system.

Soil factors involved in renovation of waste water are in relative order
of importance: 1) Secondary minerals (i.e., those minerals with a new
chemical structure due to the action of weathering on primary minerals);
2) organic matter in combination with Al, Fe or other metals; 3) uncombined
oxides, carbonates and sulfates that may contain Ca, Mg, Fe, Al, Mn or
S, and 4) primary minerals, in general confined to the coarser fractions
of the soil. Uncombined oxides and carbonates exist in a range of size
fractions, both as precipitates and coatings on primary and secondary
minerals. Organic matter exists in all stages of decomposition and sizes

- 75 -

both as entities and coatings or in chemical combinations with the mineral phase.

Soil Chemistry of Renovation: The ion exchange phenomena suggests an electrical balance where cations or anions are retained in the soil due to an electrical charge. The phenomena of cation exchange has been extensively studied, as compared to phenomena of anion exchange. In either case the soil system must stay in an electrical balance, thus, if cations are exchanged in the soil by addition of waste water then an equivalent electrical charge of cations must be released from the soil system. Many experiments have shown that there is no single universal order of the replacing power of cations on the exchange capacity. Theoretically, the replacing power of alkai ions is in the same order as the lyotropic series:

$$Li < Na < K < Rb < Cs$$

In a like manner the alkaline earth ions follow:

$$Mg < Ca < Sr < Ba < La < Th$$
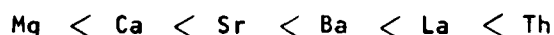
These results have been confirmed experimentally on certain clays. The replacing power of common soil cations is,

$$Na < K < Ca < Mg < NH_4$$
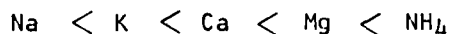
however, this order is also influenced by chemistry of the soil solution and the replacing cation (Kelley, 1948).

The charge on clay minerals of the three-layer type is due primarily to isomorphous substitution. This results in a constant negative charge density on the surface of the crystal that is not affected by the composition of the outside soil solution. In addition to isomorphous substitution, ionized SiOH and AlOH groups of two-layer clay minerals and FeOH or AlOH from coatings and gels may adsorb both cations and anions. Thus, it is apparent that many of the exchange properties of the soil are due to contributions from oxide coatings.

Organic matter also provides a reactive surface which both adsorbs cations in exchangeable positions formed by COOH and OH groups and also may complex such ions as Fe, Mn, and even Ca and Mg which in turn may adsorb anions such as phosphate. This adsorption and complexing of minerals can

be appreciable; indeed, in very sandy soils organic matter may be the factor determining renovation ability. The cation exchange capacity of the humic acid approximates 250 to 400 meg/100 g, which is three-fold greater than that of the montmorillonite-type secondary clays and 30 to 100-fold that of the kaolinite type.

Concentration of cations such as Na, K, Ca, Mg and NH_4 in the soil solution is primarily controlled by reversible ion exchange reactions, plant uptake and the formation of insoluble solid phases such as $CaCO_3$, dolomite or $CaSO_4$. Charge and size of cations are the main factors that govern the relationship between one cation and another in the soil solid-phase solution system. As a consequence of the negative charge, the composition of the soil solution in close proximity to the clay plates differs from the composition of the soil solution outside the sphere of influence of the negative electric field. The system consisting of a net negative charge on the surface and a layer of cations in close proximity to the surface of the clay gives rise to the formation of an electric double layer. The extent of the double layer in a given soil is a function of the valency of the cation and the total salt concentration in the ambient solution. Monovalent cations will be bound less than divalent cations and will thus swarm out farther away from the clay plates than divalent cations. The lower the salt concentration of the ambient solution, the further away the double layer will extend. Although the interaction of cations in soils is mainly governed by charge effects, specific forces are seldom absent. Thus, the strong and sometimes essentially irreversible binding between cations and the solid phase may be due to a high binding energy of the bond or to mechanical factors which prevent the movement of ions once they are bound. Potassium and ammonium "fixation" by illite clay minerals is probably the result of both electrostatic binding and mechanical hinderance.

The quantitative description of the solid-phase soil-solution relationships of cations can be generalized, but that of anions require, in general, individual treatment. Those anions that are bound by the soil generally reflect both a sharing of electrons and covalent bonding in contrast to the electrovalent bonding of the cations. The concentration of anions in soil solution is affected by the nature of the solid phase. The overall

- 77 -

negative charge of the adsorption complex causes anions to be repulsed
or "negatively adsorbed" in close proximity to the charged sites. Anion
binding is the most important phenomena which determines the soil solu-
tion concentration of anions. Different anion binding mechanisms which
are affected by pH and salt concentrations are operating in the soil.
Anions such as NO_3, Cl and SO_4 do not normally react with the soil solid
phase, and their concentration in soil solutions at any instant is
simply determined by the total amount of nitrate, chloride or sulfate
present divided by the total amount of water present, except at low
moisture contents where negative adsorption may become important. Re-
tention of SO_4 may be pronounced in the case of acid soils high in Fe
and Al oxides. Phosphorus as orthophosphate reacts with practically
all soils with an almost quantitative removal from solution. This is not
surprising as soils characteristically have very reactive surfaces
containing Fe, Al, Ca, all of which form very insoluble phosphates.
Acidic conditions favor the Fe-P and Al-P and alkaline conditions favor
the Ca-P retention. Retention of organic P compounds at colloidal sur-
faces may also take place and alter their chemical and biochemical
stability. The mechanisms of P retention are controversial. The reten-
tion of P by Al and Fe apparently involves both precipitation (reaction
with Fe and Al compounds) with adsorption predominating at low P con-
centrations. Mechanisms of retention by Ca may be similar.

Certain reactions may lead to production of N gases and thereby constitute
a possible pathway for N losses from soils. Both inorganic and biological
reactions may lead to the volatilization of N gases, however the contri-
bution of chemical denitrification to soil N losses is not well documented.
The inorganic reactions include decomposition of HNO_2 under acidic condi-
tions to give NO which may be partially converted to NO_2 and HNO_3, and
Van Slyke type reactions of HNO_2 with amino acids or NH_4 to yield N_2.
Such reactions may partially account for N losses which cannot be attri-
buted to biochemical denitrification.

Plant uptake of P and N combined with biological and chemical immobiliza-
tion of P and volatilization of N through biological and/or chemical deni-

- 78 -

trification, escape of NH_3 from alkaline soils, and NO_3 leaching provide the main mechanisms **for** N and P removals from soil solution.

The insoluble hydrous oxides of Mn and Fe, which are nearly ubiquitous in soils provide one of the principal controls on the fixation of heavy metals (micronutrients), such as Ni, Cu, Co and Zn, in soils. The common occurrence of these oxides as coatings allows the oxides to extend chemical activity far out of proportion to their total concentrations (Jenne, 1968). The principal factors that control the sorption and desorption of the hydrous oxide occluded heavy metals are redox, pH, concentration of the metal of interest, concentration of competing metals, concentration of other ions capable of forming inorganic complexes and organic chelates. Of these factors, pH and redox are probably the most significant. Other suggested mechanisms of control of the soil solution concentrations of heavy metals include: 1) Sorption by the layer silicates through surface sorption, surface complex ion formation, lattice penetration, and ion exchange; 2) metal fixation by organic matter; 3) surface sorption or surface precipitation on carbonates, and, 4) precipitation as the discrete oxide or hydroxide. With micro-concentrations of heavy metals sorption reactions may predominate while precipitation frequently occurs with micro-concentrations. Several of the sorption mechanisms of heavy metals are proposed to account for the fixation of metals in the sense that they cannot be extracted with salt solutions.

Soils are usually considered to have a finite capacity for cation exchange. This is frequently measured using normal ammonium acetate in application of the law of mass action. Recent studies (Bourgeois, Personal Communication) indicate a dynamic interchange between the colloidal chemical complex and availability of cations on exchange sites. Under certain conditions soil colloids will sustain an exchangeable release of potassium as potassium is sequentially stripped from the exchange sites. There is no reason to believe that this is not a reversible reaction and potassium can saturate the exchange sites only to reach a new equilibrium by fixation of significant amounts of potassium. There is also no reason to believe that many other cations will not function in a like manner.

Carbon-Nitrogen Ratio (C:N): The balance between the percentage of carbon and nitrogen in soils can have many significant influences on soil properties and processes. The carbon-nitrogen ratio is known to be important in the rate of decomposition of organic materials as well as the supply of available nitrogen to plants. If the ratio of carbon to nitrogen is high (20:1) microorganisms compete more successfully for nitrogen compounds than higher plants, thus inducing nitrogen deficiencies in plants. Under conditions of high amounts of carbon, substantial quantities of nitrogen will accumulate in the soil in association with microorganisms for the decomposition process as waste water is applied. In agricultural soils rates of fertilization attempt to maintain carbon-nitrogen ratios near 10 so that nitrogen is never limiting to plant growth. Under these conditions high amounts of nitrogen (particularly in the nitrate form) would be water soluble in the soil profile. In forest soils the carbon-nitrogen ratio of freshly needle and leaf material usually is 30-40:1. As decomposition progresses these ratios drop to 20:1 in the surface soils (0-3 inches) and 10-15:1 deeper in the soil profile (12-48 inches).

Forest soils in 100-inch precipitation zones in the Pacific Northwest accumulate as much as 10 tons per acre of total nitrogen in the rooting zone. These ecosystems are typified by large annual depositions of cellulose and lignin which occur in varying states of decomposition in the soil. Moisture and temperature conditions are favorable for year-round decomposition at low elevations. As moisture or temperature becomes limiting to microorganisms, accumulations of carbon increase and total nitrogen decreases with changes in soil formation processes.

There is a definite possibility that nitrogen renovation could be substantially improved by maintaining high carbon-nitrogen ratio (20:1 or greater). This could be accomplished by adding carbon in turning forage crops under or application of wood waste or other materials high in cellulose or other forms of organic carbon at the same rate as to total nitrogen in waste water.
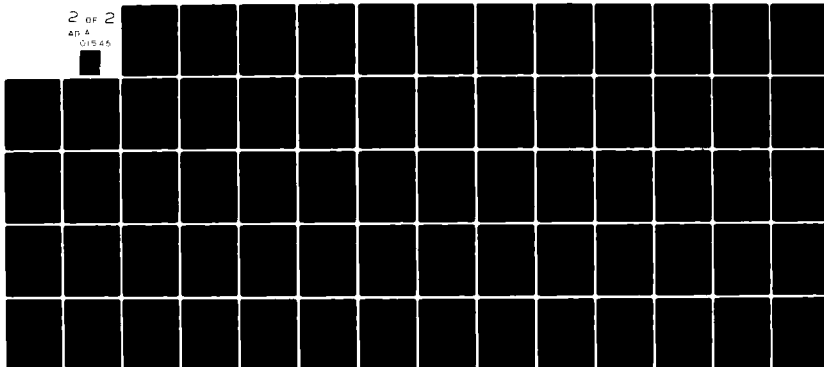
Secondary Mineral Formation: The study of soil development commonly
assumes that changes in size of primary parent materials is one of
successively smaller constituents due in part to physical reactions,
thermal expansion, contraction, or freezing and thawing, and in part
by chemical reactions involving the breakdown of crystalline structure
induced by absorption of water or organic substances. The chemistry
of the soil solution is such that secondary minerals will crystallize
in a series of structural units over time. Aluminosilicate minerals
are aggregates of oxygen ions, the interstices occupied by cations
(Al, Si, etc.) that hold the structure together. The size of the inter-
stices determines the size of the cations that can occupy them. The
stability of the crystal structure depends largely on how well the cations
fit these interstices. In the formation of secondary minerals the
most important initial structure is the primary formation of alumino-
silicates.

Important structural units of the aluminosilicate minerals are the silica
and aluminio tetrahedrons. Their linkage, which forms the framework of
the minerals, has been used as a basis for classification of alumino-
silicates. The classification leading to the formation of secondary
minerals is based on the method of linkage of the tetrahedrons as follows:
1. Independent tetrahedrons sharing no oxygen with neighboring
   tetrahedrons. Tetrahedrons are linked together with base cations,
   orthosilicate-type structure. Examples are olivine, zircon and
   garnate.
2. Separate tetrahedrons sharing one or more corners with neigh-
   boring tetrahedrons.
   a. Si_2O_7, double tetrahedron
   b. Five tetrahedrons in open groups
   c. Rings of three tetrahedrons
   d. Rings of six tetrahedrons
3. Infinite chains of tetrahedrons, metasilicate type.
   a. Single chain linkage, pyroxenes
   b. Double chain linkage, amyphiboles

- 81 -

4. Infinite sheets of tetrahedrons; two-dimentional linkage, talc,
    micas and clay minerals.

5. Continuous framework of linked tetrahedrons sharing all four
    oxygen atoms with neighboring feldspars, zeolites and quartz.

The structural order of crystallization of secondary minerals follows:
Independent tetrahedrons single chains, double chains, sheets. In this
formation process the most basic minerals crystallize first followed
by minerals with a decreasing degree of basicity. This order starts with
minerals such as olivine through augite, hornblende, biotite to quartz.

Important factors affecting the linkage of aluminosilicates and formation
of stable tetrahedrons are: 1) The presence of ferrous iron or other
cations that may oxidize during weathering as some other cations will
leave the structure to maintain electrostatic neutrality. Departure
of such cations weaken the structure; 2) Tightly packed oxygens about
a cation in the tetrahedron position occupy a smaller volume thus, are
more stable; 3) Closely related to the tight packing of oxygen is the
occurrence of empty positions (isomorphic substitution) in certain parts
of the structure. Empty positions reduce the electrostatic forces that
bind the structure; 4) As the crystalline structure of a secondary
particle grows it becomes more stable forming double chains and sheets
which have reduced surface area exposed to chemical reaction; 5) The
relative stability of a given secondary particle is also affected by the
nature of chemical organization, structure and mineral associations.

The reactions considered in the formation of secondary minerals include
ion exchange, hydrolosis, defusion, oxidation reduction and absorption.
Several important factors are related to the occurrence of these reactions:
1) Water is essential in all of these reactions, it functions as the
solvent and is the most important source of H^+ and OH^- ions. Water is
also the most important polar liquid involved in absorption, swelling
and gelation; 2) The reactions may occur in a random order, thus appearing
as one reaction; 3) Different minerals might occur due to a variation in
the order of reactions; 4) A given mineral can be viewed as resulting
from any one of several reactions. For an example, K biotite may yield
K^+ either as an ion exchange reaction or as a result of hydrolosis.

In considering the growth or breakdown of secondary minerals, the weakest bond in each structure determines the degree of stability of the whole structure. A cation that fits well in the interstices develops a stronger linkage to the surrounding structure than an ion which is either too large or too small. The weakest bond is often the most important feature in a crystalline structure.

Growth of the crystal lattice of secondary clay minerals is very slow under natural conditions. The process involves chemical weathering of the primary minerals with recrystallization and growth into secondary minerals. For example in well developed soils of the mid-West, clay has formed in the surface 0-6 inch layer at a rate of 20 grams per 100 grams of parent material in the last 10,000 years. This is an annual rate of formation of 0.0002 grams per 100 grams of parent material. The surface area of these secondary minerals in calculated to be 16 square feet. This rate of formation under natural conditions is only two and a half pounds of clay per acre per year. Given the conditions of waste water disposal and a substantial increase of base minerals in a silicate system, clay formation (secondary minerals) could proceed at an accelerated rate. The rate is assumed to be controlled by the nature of the adsorbing mineral surfaces and the supply of reactant chemicals. Waste water disposal by spray irrigation would provide sufficient water supply with H^+ and OH^- ions along with a continuous hydrolosis and the essential soil solution for clay mineral formation. If processes of secondary mineral formation could be accelerated through additions of needed elements to waste water, it is possible that certain soils might have an infinite capacity over years for renovation of the waste water through development of new soil particles.

Available Nutrients in Relation to Plant Uptake: Some degree of renovation of waste water in soil systems may be expected from the uptake of nutrients from the soil solution by plants. For renovation to be complete, the bulk of the plant crop must be harvested and the nutrient supply removed from the ecosystem. Table IX-1 shows the annual uptake of nutrients

- 83 -

Table IX-1

ANNUAL UPTAKE OF ELEMENTS BY FOREST AND
CROPS IN RELATION TO QUANTITIES
APPLIED IN WASTE WATER*

Element	Waste Water	Forest	%	Crops	%
N	470	23-60	12	160	34
P	235	6-20	8	25	10
K	328	3-67	20	167-279	85
Ca	565	6-53	9	70	12
Na	1,175	1- 3	0.2	99	8
S	--	4			
Fe	10	1	10		
Mn	5	1	10		
Cu	3	0.1	3		
Mg	398	5- 8	2	29	7
C	2,000+	3,500			

*Units in lbs./acre.

by forests and agricultural crops in relation to the quantity supplied in the model waste water effluent. Potassium has the highest turnover rate and the greatest annual uptake of any element (forest, 20% and crops, 85%). A forest would take up only 12% of the total nitrogen applied as compared with 34% for agricultural crops. These data make it apparent that harvesting of crops will not approach a complete solution for renovation of waste water for important macronutrients. The forest is probably much less efficient as the bole is the only portion which is ultimately harvested and it contains the lowest percentages of macronutrients, N, P and K.

Few data exist which compare exchangeable nutrients with the total quantity in a soil. Table IX-2 shows the relationship of exchangeable ions to the total concentration in the surface two inches of forest soils. This depth (0-2 inches) is the most chemical reactive portion of the forest soil but is also the zone of maximum leaching. Even at this depth, however, there is a reserve supply of elements from 100 to 600 times the amounts available as ion exchange. These data suggest that the soil exchange capacity might establish a new chemical equilibrium with additions of elements in waste water to the soil solution.

Sopper (1971) has shown that renovation of a waste effluent, very similar to our model effluent, can be achieved for many elements in a 4 foot soil profile. Nitrate was reduced 82%, total organic N, 93%, phosphate, 99%, potassium, 83% calcium, 59%, magnesium, 53%, manganese, 66% and boron, 68%. There is no renovation of chloride and sodium added to the soil. Sopper's work, however, also shows no significant differences between the water quality of percolate 4 feet deep in the soil profile for organic N, P, K, Ca, Mg, Mn and B. Again, only chloride and sodium show significant increases following waste water application. Both of these elements occurred in percolate in higher concentrations than in the effluent applied.

B. Inorganic Chemical Interactions

Exact inorganic chemical interactions with application of waste water to soil systems cannot be predicted precisely. The assumption that a chemical

Table IX-2

RELATIONSHIP OF QUANTITIES OF IONS ON THE
SOIL EXCHANGE CAPACITY TO TOTAL AMOUNT
IN THE SURFACE SOILS WITH RATIO OF
TOTAL TO EXCHANGEABLES

Ion	Soil Exchange Capacity meq/100 g. O.D. Soil	Total	Ratio Total/ Exchange
Ca	0.60	110	183
Mg	.73	93	127
K	.10	24	240
Mn	.009	.44	490
Na	.14	92	657
Zn	.009	---	
Fe	---	99	
Al	---	767	
Total Exchange Capacity	17		
P in mg/l	43	1,095	25

interaction will take place is based on passage of the waste water effluent through the soil system. This is achieved only with spray irrigation and rapid infiltration ponds. The coarse, porous soils required for rapid infiltration ponds suggests renovation to a lesser degree than through spray irrigation. There should also be marked differences between the five study areas as soil chemical reactions are a function of climate, soil parent materials and vegetation. The following sections summarize the expected chemical interactions of macronutrients and heavy metals.

Nitrogen and Phosphate: In general N removal varies from less than 0 to 95% removal depending primarily on soil type and depth of soil column, design and mode of application of waste water and vegetation cover. The P removal varies from less than 0% to 99% and is governed primarily by soil type and depth of soil column. The specific effects of the most important operating factors are briefly discussed hereafter. (Table IX-3)

The soil type will have a large influence on the N removal. An increasing clay content will increase soil denitrification due to a more favorable soil structure. On the perimeter of soil aggregates aerobic conditions are present while inside the aggregate the nitrate under low O_2 pressures can be reduced and diffused into the air as N_2 or N oxides. The depth of the soil column should be great enough to give a minimum layer of 2 feet in which nitrification and denitrification takes place. The P (Figure IX-1) removal will increase with the clay content of the soil or concurrent, decrease of the permeability. The P is generally effectively removed in the upper 1 to 2 feet of the soil by adsorption precipitation reactions when clay, sesquioxides of Fe and Al and calcareous materials are present. It was calculated that every 10 years a depth of 1 foot will be saturated with P which can limit the use of certain soils. When overland runoff is practiced this may become a limiting factor over longer periods of time because runoff water does not penetrate very deeply into the soil.

The effect of the dosage and schedule of application and amount of waste water applied (hydraulic loading) can be described in the same way as it

is done for normal sewage treatment systems. A higher load results in a larger mass removal and a lower removal percentage. Increased loadings under certain conditions may shorten the longevity of the soil due to organic matter accumulation that cannot be degraded rapidly enough. Higher moisture contents that are the result of higher loadings will create locally anaerobic conditions that are necessary for denitrification and N removal will, therefore, increase with loading until this reaches a point where NH_4^+ cannot be effectively nitrified anymore and N removals will drop sharply thereafter. Figure IX-2 indicates the importance of the schedule of application and shows that a single dose per week creates highly aerobic conditions in the soil that do not favor denitrification. At four doses per week the N removal by denitrification can reach 60 to 90%. A higher dosage per week also increases C removal but may shorten the longevity under high loading conditions. A selected dosage will, therefore, always be a compromise between N and C removal and longevity. A 2 inch/week load of the waste water effluent considered in this report will not adversely affect the longevity of the soil.

The N removal in rapid infiltration ponds is mainly governed by the period the basin is inundated (Figure IX-3). A longer period gives a better balance between aerobic and anaerobic conditions necessary for nitrification-denitrification. However, prolonged inundation periods, which create permanent anaerobic conditions that inhibit nitrification should be avoided.

Potassium: Potassium is a very mobile and dynamic element in both soil and plant systems. It will displace sodium from the soil cation exchange capacity, but is in turn one of the most easily displaced by other divalent or monovalent cations. Many clay minerals have a very large capacity to fix potassium by isomorphic substitution in the clay lattice. This phenomena should be important along with the incorporation of potassium in formation of secondary minerals. Long term renovation of potassium by the modal soil could be expected to average near 50%.

N AND P REMOVAL

Table IX-3 1 of 3

Irrigation System Author	Amount Sprayed	Dose	Duration	N Removal %	P Removal %	Soil Type	Clay %	Soil Cover
Rapid Infiltration Ponds								
Bouwer, 1971a,b	36 inch/day	2 day infiltr. 2 day rest	365 d/y	0-10	62	silty sand	2	bare-grass
	36 inch/day	14 day infiltr. 14 day rest	365 d/y	80	62	silty sand	2	grass
Baars, 1957a,b, 1961	55 inch/day	6 months infiltr. 6 months rest	6 m/y	92.4	92.5	sand	0	grass
Merrell, et al., 1967	14 inch/day	intermittent	——	41	92.7	silty sand	-	bare
Gotaas, et al., 1953	6 inch/day	7 day infiltr. 7 day rest	365 d/y	11.4	100	sandy loam	8-11	bare
	6 inch/day	14 day infiltr. 7 day rest	365 d/y	42.0	93.6	sandy loam	8-11	bare
	6 inch/day	continuous 120 d. few months rest	365 d/y	85.7	55	sandy loam	8-11	bare
McMichael and McKee, 1965	12 inch/day	1 dose/day 10 hour	365 d/y	0-58	0-34	sandy	2	bare
Overland Runoff								
Bendixen, et al., 1969	0.9 inch/day	2 doses of 6 hrs. 5 days/week	150 d/y	90	65-84	clay loam	35	grass

Table IX-3 2 of 3

N AND P REMOVAL (Continued)

Irrigation System Author	Amount Sprayed	Dose	Duration	N Removal %	P Removal %	Soil Type	Clay %	Soil Cover
Overland Runoff (Con't.)								
Law, et al., 1969, 1970	0.63 inch/day	1 dose of 6-8 hr. 5 days/week	301 d/y	84	56	sandy loam	10-35	grass
	0.63 inch/day	1 dose of 6-8 hr. 3 days/week	301 d/y	84	83	clay loam	35	grass
Spray Irrigation								
Bendixen, et al., 1968	8 inch/day	1 dose/day	7 d/w	30	10	sand (lysimeter)	0	bare
Sopper, 1971a,b	2 inch/week	1 day/week	Apr-Nov	80	99	loam	15-35	forest
	4 inch/week	2-3 day/week	Apr-Nov	91	99	loam	15-35	forest
	2 inch/week	1 day/week	Year round	0	95	loam	15-35	forest
Foster, et al., 1965	13.4 inch/week	3 day sprayed 4 day rest	365 d/y	56	91	silty sand	2	forest
Ridge and Furrow								
Bendixen, et al., 1968; Schwartz and Bendixen, 1970	1.6 inch/day	1 dose/day	7 d/w	20	30	sand (lysimeter)	0	bare
	1.6 inch/day	1 dose/day	7 d/w	50	96	silty loam (lysimeter)	15	bare

Table IX 3 of 3

N AND P REMOVAL (Continued)

Irrigation System Author	Amount Sprayed	Dose	Duration	N Removal %	P Removal %	Soil Type	Clay %	Soil Cover
Ridge and Furrow (Con't.)								
	1.6 inch/day	1 dose/day	7 d/w	85	99	silty loam (lysimeter)	15	grass
	3.2 inch/day	1 dose/day	7 d/w	70	93	silty loam (lysimeter)	15	grass

* Concentrations of N and P in the effluents used in the experiments reported in this table were around 20 and 10 mg/l, N and P, respectively. Also, all the results were obtained under field conditions, except if otherwise stated.

Figure IX-1

MAXIMUM AMOUNT OF P ADSORBED IN THE A HORIZON OF
THE SOIL AS FUNCTION OF THE SOIL CLAY CONTENT
(Ellis and Erickson, 1969)

Figure IX-2

EFFECT OF HYDRAULIC LOADING ON N REMOVAL
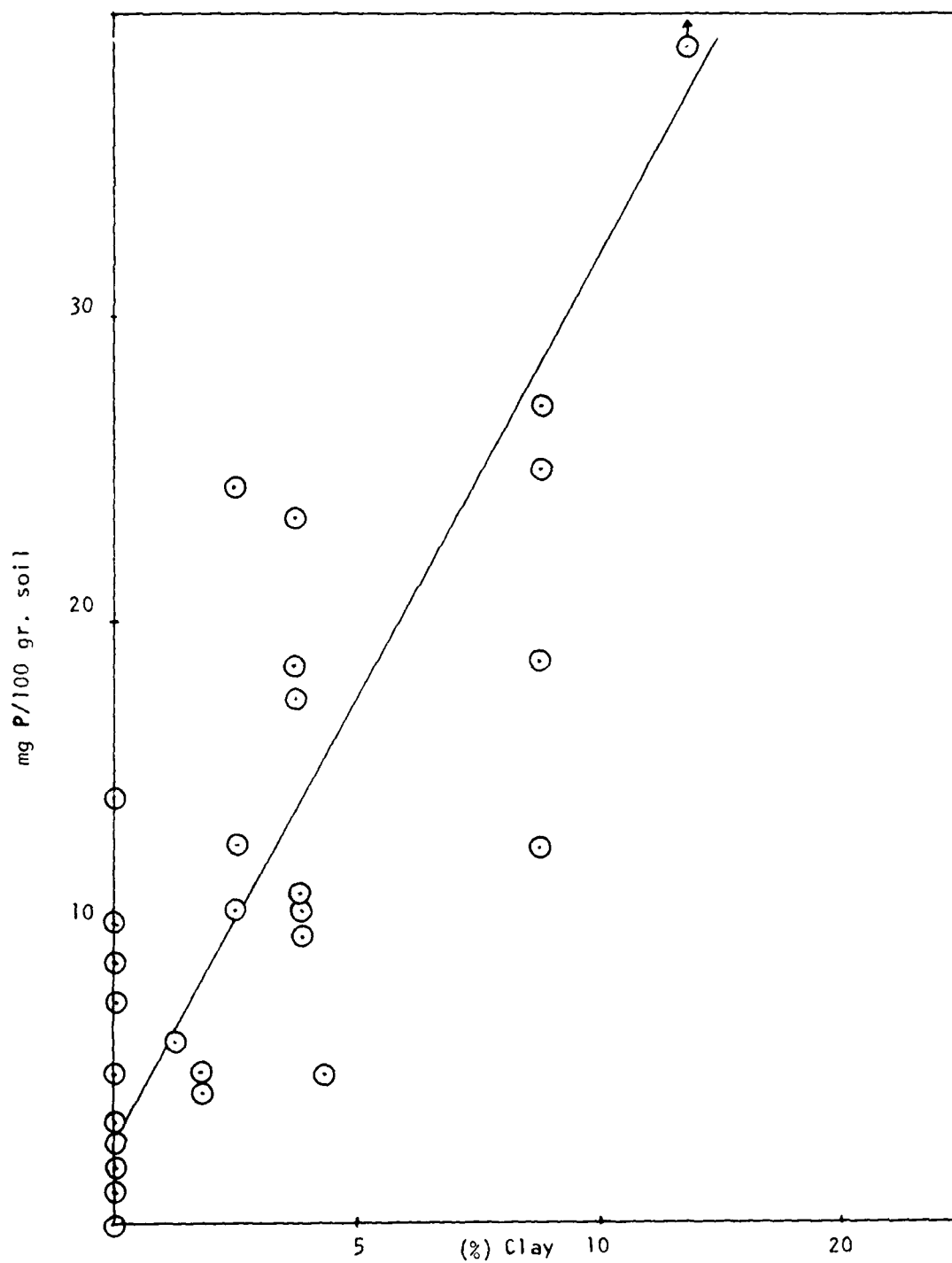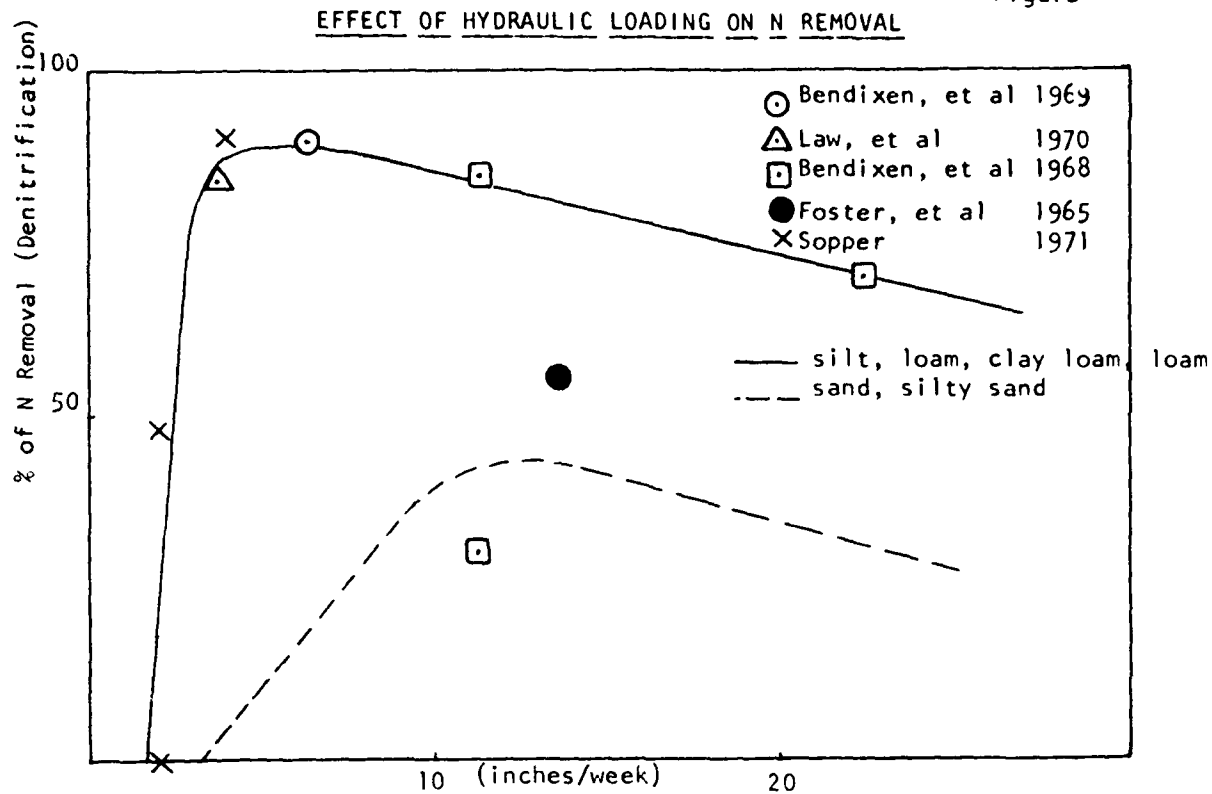
Figure IX-3

EFFECT OF LENGTH OF INUNDATION PERIOD ON
DENITRIFICATION AND N REMOVAL IN RAPID INFILTRATION PONDS
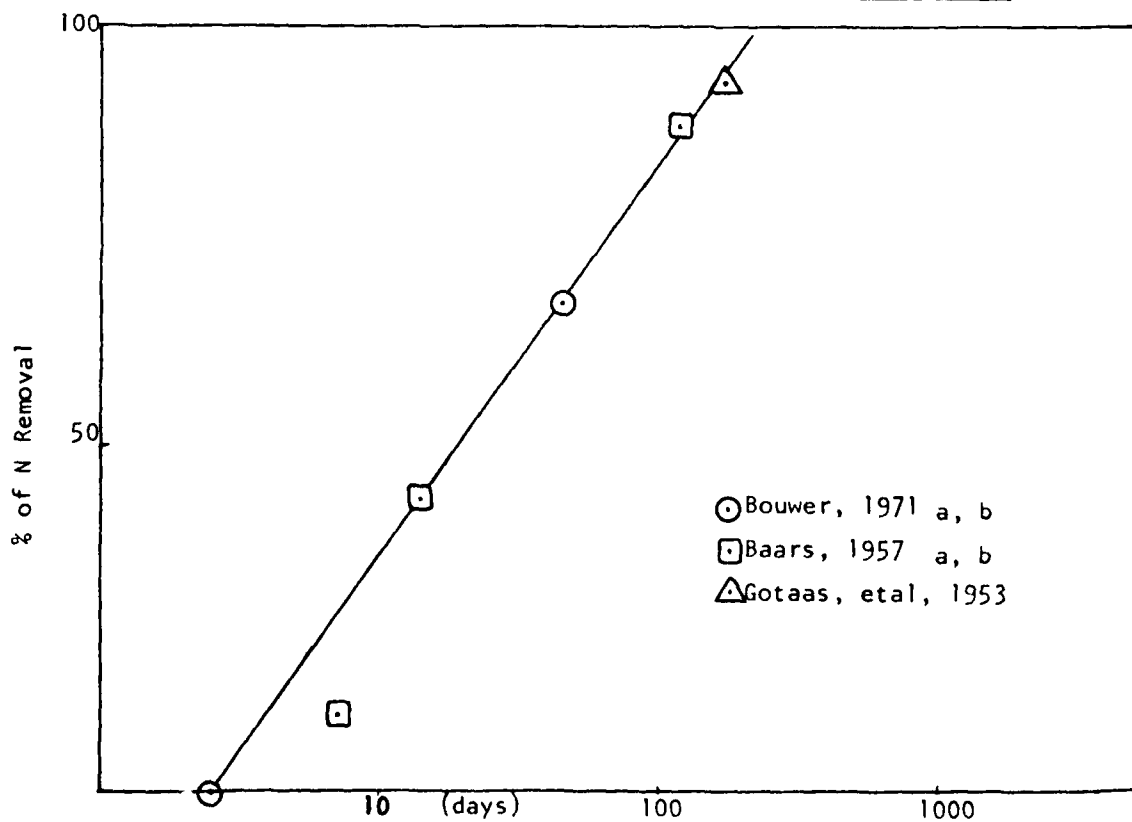
Calcium: Calcium may be involved in a series of chemical reactions in-
cluding ion exchange, precipitation and fixation. Sustained capacity
for a soil to renovate calcium loading should range from 60 to 80% for
humid regions but possibly much less for California. Higher quantities
of phosphate could lead to the formation of common apatite, Ca_5 (F, Cl)
$(PO_4)_3$ or other Ca minerals (augite).

Sodium: Is one of the major cations in waste water effluent and is not
expected to be renovated by the soil. In the California area sodium
leaching could increase substantially due to waste water applications as
greater amounts of water would flush the soil profile. Sodium would
probably be the cation which maintains electrical neutrality as chloride,
nitrate and sulfate pass the soil system.

Magnesium: Magnesium is retained with greater affinity than calcium on
the soil exchange capacity, thus would be expected to have a higher de-
gree of renovation if ion exchange is the main renovation mechanism.
Sopper's work (1971) shows decreased renovation for magnesium over calcium.
The balance between renovation of calcium and magnesium would be a func-
tion of the characteristics of the soils under consideration, particularly
balance of Ca and Mg on the exchange capacity.

Anions: Bicarbonate, chloride and sulfate are major anions which will be
very mobile in soil systems. Bicarbonate is usually considered a by-pro-
duct of respiration of soil organisms and frequently makes up the anion
needed to maintain electrical neutrality. Chlorides will not be renovated
in the soil system and sulfate possibly only to a very minor extent as it
might occur as precipitates under certain conditions.

Heavy Metals and Near Metals: Cultural practices in management of agri-
cultural soils where large quantities of irrigation water and other chemi-
cals have been added may be used as examples of the impact of waste water
loading for these heavy metals, near metals and trace elements.

Iron and Manganese: Fe and Mn are major constituents in many soils and very active in redox processes. In well aerated soils Fe and Mn are stable oxides and would be very immobile (near neutral pH). Water logging of soils or other forms of reduced aeration would decrease pH and increase mobility of both elements. Under design loading all Fe and Mn would be renovated in California soils and 90 to 95% in Lake States and New England soils.

Arsenic: Webster (1956) reviewed the history of insecticide applications on deciduous fruits in Washington State. Applications of arsenate were first begun about 1903. Rates of application before 1920 were moderate. Few areas would have received as much as 100 pounds of As per acre in total. In the period 1920 to 1948 an average orchard received about 50 pounds of As per acre per year or 1400 pounds total. Many orchards received much more.

In the period before 1946 almost all fruit in eastern Washington was irrigated by the rill or furrow method (Overly, 1950). This method of water application does not facilitate leaching from the surface six inches of soil. Since the fruit areas receive only about 8 inches of precipitation annually there was little removal of As from the surface six inch layer until sprinkler irrigation was employed. Soil analysis during the 1940's and 1950's showed very high arsenic concentrations. Typical analyses are those of Walker who analyzed a number of field plots at the Tree Fruit Experiment Station for F. L. Overley (1943). He found a range of 99 to 1179 mg/l with the median value about 650 mg/l. Values below 500 represent pear orchards or land out of orchard for several years. Similar values were found by Benson (1953) who collected soil samples from 17 orchards in North Central Washington. He found the total arsenic to range from 108 to 830 mg/l with the median about 450.

In the period following World War II there was a general change in the irrigation method employed by fruit growers. Rills were discontinued in favor of sprinkler systems. Soil sampling to a depth of 4 feet (Thomas, 1955) showed that there was a considerable amount of arsenic in the second 6 inches

of soil and significant amounts in the second foot. Benson[1] sampled
62 representative orchard soils in 1969. These showed that a significant
change in soil arsenic distribution had occurred. After about 20 years
of sprinkler irrigation with an average annual application of three feet
of water and no additional arsenic the As concentration in old orchard
soils ranged from 20 to 456 mg/l in the surface foot. It is to be re-
cognized that samples taken to one foot would show a reduced concentra-
tion if no downward movement had taken place because of the dilution
effect of the second 6 inches of soil. However, the soils showed an
average arsenic concentration of 113 in the first foot, 45 in the second
and 24 mg/l As in the third foot depth. Most soils had at least 50% as
much in the second foot as in the first and again 50% as much in the
third as in the second. A few soils with high arsenic values in the
surface foot had relatively low values in the subsoil. This could re-
sult from rill irrigating over most or all of the past 20 years. Even
so, average values indicate that arsenic has moved into subsoils and
is continuing downward.

Thomas (1955) concluded that the soil arsenic was still associated with
lead. He thought that residual arsenic existed as iron or aluminum lead
arsenate complexes. Woolson(1969), working with soils of much broader
chemical, physical and mineralogical composition, concluded that soil
arsenic reacts with iron and aluminum hydroxides and gels. When arsenate
combines with iron compounds in the soil it forms complexes that are un-
available to plants and resists further change (Jacobs, et al, 1970).
Woolson concluded that most soils in the U.S. contain sufficient iron and
aluminum in reactive form to detoxify large quantities of arsenic. "Large"
would mean amounts equivalent to the heavy applications already made.
Exceptions are the arid areas of the West where there is not sufficient
active iron to "fix" the arsenic. To substantiate Woolson's conclusion
it can be noted that the red and yellow soils of the Southeast exhibit
no arsenic toxicity even though large applications of calcium arsenate

_1/ Unpublished

- 96 -

were made to cotton in the day prior to DDT. Similarly, fresh additions of arsenic to a forest soil from White Salmon, Washington show no toxicity when 200 mg/l were added.

Woolson (1969) found reasonably good correlations between dilute acid, soluble arsenic and plant growth. His value was $r = 0.74$. An $r = 0.64$ was found for Washington soils between corn and dilute acid soluble arsenic.

Plants do not absorb significant quantities of arsenic from soils even though the soil may contain large amounts. Arsenic is primarily toxic to roots of plants and is not translocated upward to a significant extent. An exception is the peach. Lindner (1943) found that peach leaves show symptoms when they accumulated 4 mg/l As. Normally foliage of plant growing in arsenic contaminated soils will contain 5 to 10 mg/l As and occasionally will have up to 20 mg/l. Some plants show little if any toxicity to soil arsenic in concentrations up to 2,000 mg/l. This is true of most grasses.

Lead: Calculations on the amount of lead deposited on eastern Washington orchard soils can be made from the same information as that for arsenic given earlier. The amount would be roughly 3 times that of the arsenic. Overley (1950) calculated that 3,222 pounds of lead would be deposited in 10 years. Few analyses for lead in eastern Washington soils have been made. Jones and Hatch analyzed soils in 1937 and found up to 1,383 mg/l total lead in surface soils. They concluded that lead was not being transported downward but their analytical data does not entirely support that conclusion. One Washington soil extracted with dilute acid showed 35 mg/l in the surface and 1 mg/l in the next 2 feet. This indicates that most of the lead has reverted to nonexchangeable form and that some downward movement has taken place.

Lead is being added to soils from auto exhausts in significant amounts near high traffic density highways. Several studies have been made to determine the extent of the disposition and its movement in soils (Lagerwerff, et al, 1970, Motto, et al, 1970, Page and Ganje, 1970). Typical data is that of Lagerwerff and Specht who found Pb soluble in

1NHcl to be 522 mg/l in 0-5 cm, 460 in 5-10 cm and 116 in 10-15 cm. This indicates some movement but sampling was of limited depth. Page and Ganje have data for soils taken in 1919-33 and in 1967. They found 2 to 3 times as much Pb in 1967 where traffic was heavy but no change where traffic was light. They also show measurable increases in lead in the 2.5 to 15 cm depth but no movement to lower depths. Likewise, Motto, et al (1970) showed increased levels at the surface but no measurable increases below 10 inches. It is generally agreed that lead may enter into exchange but fixes to nonexchangeable forms, some of which is dilute acid soluble (Keaton, 1937, Swaine, 1958). Some leaching or downward movement takes place but it is slow. Soils developed on shales high in lead show some loss from the upper profile, e.g., the Ae horizon had 752 mg/l compared to 1,431 in the C (Present and Tupper, 1965).

Plants do not transport appreciable amounts of lead to the upper parts. The roots may contain several hundred mg/l but leaves rarely contain more than 10 (Motto, et al, 1970) Higher analyses for leaves indicate deposits from aerial sources.

In conclusion, lead is fixed in soils and not readily mobilized. Soils should retain several hundred mg/l against leaching.

Copper: Excess copper use has rarely occurred in agriculture despite the fact that copper is a commonly employed fungicide. The largest area of copper toxic soils occurs in sandy citrus soils of Florida (Reitz, et al, 1964). These soils have extremely low clay content and a base exchange capacity of only 2 to 4 meq/100 g. The C.E.C. resides in the soil organic fraction. Copper forms a tightly bound complex with the organic matter but when the total copper content exceeded 50 mg/l in the soil toxicity to citrus and other plants occurred. The copper has remained in the surface soil. These soils are easily detoxified with lime by maintaining soil pH above 6.5.

In apples banic copper or copper-lime mixtures have been used to control a root or crown rot fungus. The trunks of trees are sprayed with a solution

containing 3 pounds of Cu per 100 gallons. To determine the toler-
ance of apple trees to copper, Covey and Benson established a pot
culture test with 0, 5, 50 and 500 mg/l Cu on a soil acidified to pH
4.5, not acidified, pH 6.0 and limed with 10% free lime. Copper toxicity
occurred only with 500 mg/l Cu on the acid soil but not on other soils.

Copper moves very slowly in soils, probably as an organic complex. Soils
with a high native copper lose copper in development. Present and Tupper
(1965) showed the Ae horizon to contain 32 mg/l while the B_2 had 223 mg/l.

Copper in soil is not transported to the aerial parts in high concentra-
tion. Normal values are 5 to 15 mg/l. Higher values usually indicate
surface contamination. Roots may accumulate much more.

Zinc: Most soils contain limited amounts of zinc so that deficiency is
a common occurrence. Applications of zinc are usually in the order of
15 to 30 pounds per acre. This amount rarely remains available more than
3 to 5 years when repeat applications are necessary. For tree fruits
and nuts, applications of a ton per acre usually does not suffice to
overcome zinc deficiency. Numerous studies have been made to determine
the cause of poor uptake of zinc by trees and grapes. The usual ex-
planation by horticulturists is soil fixation but such an explanation is
not adequate.

Benson (1965) and Sharpless et al (1969) studied zinc retention by several
soils when sufficient zinc was added to saturate the soil. Arid soils
retain about 80% of the zinc as exchangeable. The fixed zinc is partly
acid soluble and available to plants but some is fixed as zinc silicate.
Zinc may be removed from exchange position by other ions especially
hydrogen. Percolating solutions of salts will replace zinc and carry it
deeper into the soil column. Alkaline or calcareous soils will retain
large quantities of zinc well in excess of the cation exchange capacity.
The zinc is precipitated as zinc oxide or carbonate. Movement of zinc
in calcareous soils is difficult.

- 99 -

Soils that have become toxic due to a high quantity of zinc may be
detoxified by maintaining the soil pH above 6.5. This has occurred
in sandy coastal plain soils of the Carolinas which received sprays
containing zinc on peach trees. When the trees were removed other
crops failed to grow normally unless the land was limed.

Other Toxic Materials: Fluoride is not absorbed from soils unless the
pH is very low, below 4.5. An unpublished experiment by Reynolds
Aluminum Company near Portland showed Gladiolus to be severely injured
from soil absorbed F in extremely acid soils but healthy plants grew
when lime was applied. Fluoride is adsorbed by soils to a greater ex-
tent than most anions but it also forms the highly insoluble CaF_2.
This compound is not toxic to plants unless disassociated.

Cadmium and nickel are components of auto exhaust. They are emitted
in very small amounts but some accumulation does occur near highways
of heavy traffic density (Lagerwerff and Specht, 1970). Accumulation
occurs in the surface soil. Leaching should be limited.

Most heavy metals can be expected to concentrate in sludge and sediments
(Toth and Ott, 1970). Very low concentrations should occur in the
effluent especially if the pH is neutral. The concentration of toxic
elements expected in treated sewage effluent contribute very small
quantities per acre per year and should not create problems for several
hundreds of years.

C. Mechanisms of Renovation - Organic Compounds

Metabolism by soil microorganisms is by far the leading route by which organic compounds are removed from waste water. A very wide variety of organics may be handled this way. Examination of the classes of organics shows they are not what would be considered energy rich and generally their metabolism will initially result in minimal cellular production. Ultimate degradation to CO_2 will occur unless non-ideal conditions such as anaerobic conditions, insufficient contact time and overloading exist. Minor losses of certain very volatile organics may occur directly to the atmosphere, especially with spray irrigation. Chlorinated hydrocarbons are lost by volatilization, but will not be removed from the environment.

Some larger molecules may be retained in soil by filtration, providing further time for microbial action. This is a very minor route for renovation. A more important mechanism is adsorption. This is especially important with polynuclear hydrocarbons and with chlorinated hydrocarbons, infact is about the only effective method for removal of these classes of compounds. This route is not important with other classes of compounds.

Plant uptake can theoretically occur with a number of the classes of compounds, however, the rate compared to uptake of inorganics is very slow and little renovation occurs by this means. Few of the organic molecules are charged and can enter into ion exchange phenomena, and this too is a minor factor in waste water renovation as far as organics are concerned.

D. Organic Interactions

Classification of Probable Compounds in Secondary Effluent: Organic compounds in secondary treated effluent are not known nearly as well as inorganics, they are usually described only by BOD, COD and, more recently, TOC values. The BOD values used as a baseline, 25 mg/l, and the COD value, 70 mg/l, indicate low amounts of organics. It may be that these values are on the low side when one considers industrial sources. For example, present regulations require pulp mills to limit BOD discharge after secondary treatment to much higher values.

In developing the classification of probable organic compounds in
secondary effluent emphasis was placed on the larger sources such as
domestic sewage, the chemical and allied industries, the pulp and paper
industry and the petroleum industry. Of these, the latter two are of
much less importance than the former in the five study areas. Their
inclusion has not, however, required much extra effort and has only
slightly expanded the classes of compounds considered.

The best information is available on secondary treated domestic sewage;
here, quantitative data on classes of compounds and on some individual
compounds has been reported. Qualitative data is somewhat incomplete.
The information on the chemical and allied industries is very limited.
Some reports on new secondary treatment installations which solve unique
problems are available and, in the absence of better information, the
results of river water and sediment analysis were used to fill out what
might be found in effluent. The petroleum industry generally discharges
limited classes of compounds which are qualitatively described. A large
body of information is available on the pulp industry from inplant analysis.
This is the result of years of research aimed at utilization of waste
material, understanding pulping reactions, etc. Adequate qualitative
data also exists to classify the organics which survive secondary treat-
ment from pulp mill effluent.

It is very important to recognize that a summation of the individual com-
pounds reported from these various sources may present a very distorted
view of the situation. Many effluent streams have never been analyzed,
while a few have been very thoroughly analyzed. By reducing the available
data to classes of compounds, the rather scarce information can be smoothed
out to a considerable degree and undoubtedly gives a better overall view
of what might be expected in a mixed domestic-industrial effluent.

The humic substances, humic acids, fulvic acids and hymathomelanic acids,
are all present in secondary effluent and represent about 50% of the COD
of domestic effluent. These compounds will be present in waste water,
perhaps as high as 40 mg/l in concentration (Rebhun, 1971). Of this about

half is humic acid and equal amounts of the others will be present. Poly-
phenolics such as tannins are present at about the 1 mg/l concentration.
Industrial waste water may contain much higher concentrations of lignins.

Amino acids have not been reported in secondary effluent and, while they
are probably present, the concentration will be very low. Protein is
reported in secondary effluent up to 7 mg/l; this value is, however, a
nitrogen value arrived at by the kjeldahl method and probably represents
mostly non-protein material such as purines (Bunch, 1961). These survive
secondary treatment to a large extent and will be definitely present in
waste water, individually in concentrations of 10-30 mg/l

Carbohydrates are found in secondary effluent. Perhaps surprisingly glucose
is the major sugar reported, followed by rhamnose. (Painter, 1960)

Detergents such as alkyl benzene sulfonates are resistant to degradation
in secondary treatment and have been reported in the effluent in concen-
trations in the 10-15 mg/l range. These ABS types of detergent were largely
replaced by the more easily degraded linear alkylate sulfonates five or
six years ago; little information is available about LAS in effluents.
ABS type materials, responding to the methylene blue test, are reported in
the 5-10 mg/l range. (Sopper, 1970)

Neutral hydrocarbons can enter waste water streams from a very wide variety
of sources. Refineries normally discharge these compounds in small quan-
tities and the very wide use of petroleum products assures their presence
in secondary effluent since they are not readily degraded. A concentration
range of 1-2 mg/l has been reported in non-industrial waste (Rebhun, 1971).
A very wide variety of aromatic compounds have been reported in river water
and one can expect small concentrations of these compounds in waste water.
A surprising number of polynuclear hydrocarbons are also reported, their
concentration range must be very low.

Phenolic compounds are reported in sewage, refinery and pulp mill waste, as
well as several other treated waste streams. Phenolics of widely differing

- 103 -

structure must be definitely considered in waste water disposal. The
concentration range is very difficult to establish but perhaps 1 mg/l
is a good working figure.

Plasticizers are now being reported more frequently in waste water (Anon,
1971). These are generally phthalate esters, with the di-(2 ethylhexyl)
phthalate receiving most attention. These materials are apparently in-
adequately controlled in secondary treatment, and their occurrence in
waste water seems to be following the pattern of the chlorinated hydro-
carbons, i.e., heavy and widespread use combined with perhaps unexpected
stability leading to contamination of water sources. Concentrations of
phthalate esters ranging from zero to over 3 mg/l have been reported in
rivers near metropolitan and industrial centers.

Organic acids of every kind have been reported in secondary treated domes-
tic effluent and in many industrial sources. Total concentration of about
3 mg/l are found of acids of less than C_{10} and reports of other acids
are frequently reported in the 0.1 to 1 mg/l range. Many of these are the
product of the secondary treatment (Murtaugh, 1967).

A variety of synthetic nitrogen compounds have been reported in river
water. These are generally rather closely controlled because of odor
or toxicity problems and they should not generally be found in secondary
effluent. A few alcohols have also been reported, these seem to be of
minor importance, the same holds true for sulfur compounds although these
need to be considered in secondary effluent.

The resin acids and other terpenes are of importance in treated effluent
from pulp mills. No concentration data is available, but it is now clear
that these compounds are either not controlled at all, or are only slightly
modified in secondary treatment processes and will be in the effluent (Keith,
1969).

Chlorinated compounds, in general, are present in domestic effluent as well
as in industrial effluent. Their remarkable stability means they will not

be controlled by secondary treatment, and their widespread use, etc., means they will enter waste water in surface water runoff and by other means. Their inclusion in the list of compounds in secondary effluent is useful since they are models for any non-degradable organic compound which might be found in waste water. Adequate information on behavior of DDT in soils is available to make predictions for non-degradable compounds in general.

A complete summary of organic compounds reported in water is in the appendix of this report.

Removal of Organic Compounds in Soil-Water Systems: The classes including carbohydrates, proteins/amino acids, organic acids and purines are readily metabolized and will not be considered further.

The polymeric materials in the humic substances and polyphenolic classes have been partially degraded in passing through secondary treatment. They are normal components of many soils, especially forest soils, and are slowly degraded. Some interactions with other organics which should be noted are discussed below. These compounds may leach out and cause coloration in streams, but generally not in groundwater. Terpenoids, the resin acids and terpenes, are generally resistant to metabolism. It is clear that these compounds are modified by oxidations such as conversion of hydroxyl groups to carbonyls or introduction of hydroxyl groups during secondary treatment, but they are not usually completely utilized. In soils they do appear to be metabolized.

For example, conifer needles, which contain 5-10% terpenoids, are slowly converted into humus with no apparent buildup of resin acids and other terpenes. Several microorganisms have been isolated from forest soil which are capable of converting terpenoids to acetyl coenzyme A, assuring complete utilization. It seems safe to conclude that complete removal of terpenoids will be accomplished by soil microorganisms, although there is undoubtably some adaptability factor, and one might expect grassland to be less effective than forest land disposal.

- 105 -

Phenolic compounds such as the cresylic acids and xylenols from petroleum refineries are adequately controlled by carefully managed secondary treatment facilities, however, some types of phenol users, especially plywood manufacturers using phenolic resins, have been unsuccessful with secondary treatment facilities. Heavy loading is probably the main problem. As long as the baseline or near baseline quantities are maintained, microorganisms should have little difficulty utilizing phenolics since several very well worked out pathways are known by which these compounds are metabolized via ring cleavage. Phenolics can be toxic, and high concentrations may result in microorganism kill off. No evidence exists for long term problems, but it does seem that adequate control of concentration in secondary effluent is important.

Nitrogen containing organics are common decomposition products and are removed adequately by secondary treatment. Aromatic compounds might be expected to be detoxified by methylaton, then degraded by ring cleavage. There does not seem to be any reason to expect problems with this class of compounds in waste water disposal on soil.

Detergents of the ABS types generally survive secondary treatment, while LAS types are well controlled. Evidence from long term studies on forest irrigation has clearly established that the ABS type of detergent is not accumulating or leaching through the soil and it seems most probable the material is being metabolized. No long term problems are apparent here.

Neutral hydrocarbons in low concentrations are readily metabolized by a variety of microorganisms, usually by oxidation at C_1 involving oxygen. Aerobic conditions are necessary and the initial reaction product is an alphatic acid, readily utilized via a B-oxidation route. Branched chain hydrocarbons are metabolized at a significantly lower rate. Here again, soil microorganisms should have no problem as long as the baseline values are not greatly exceeded.

Aromatic hydrocarbons are somewhat more resistant to metabolism. They generally will become hydroxylated and then convert to catechols followed by ring cleavage. Baseline values again are the key; in the ranges reported

no problems should exist. Because of their very low solubility in water,
flooding should be avoided as it is important to obtain contact of all
hydrocarbons with soil for utilization to occur.

Polynuclear hydrocarbons present a special problem. They are quite
stable and seem to occur widely in small amounts. Their source is in-
dustrial -- coking operations, etc. They probably are not metabolized
to any extent but perhaps are held in soil by physical means.

Plasticizers also may be a problem. These often are phthalate esters of
aliphatic alcohols, which are fairly easily metabolized. One in parti-
cular, di-2-ethylhexylphthalate, seems to be resistant to decomposition
and is apparently accumulatory. It is likely that this compound would be
metabolized by soil microorganisms and this may in fact be the best
method of control. Prediction is difficult here and might be best
summarized by stating that in general no problems will be found with the
possible exception of the compound cited above, and with chlorinated
esters, such as those made from tetrachloro phthalic anhydride.

The last class of compounds is the chlorinated hydrocarbons, the very
stable group of pesticides which have received so much attention. These
compounds are expected to be present in secondary effluent in very low
concentrations and may be present in effluent from a manufacturing facility.
They are very widespread in water at present. Their metabolism is very
limited in soil and they tend to accumulate. The chlorinated hydrocarbons,
once into the soil, tend to remain there. Under certain conditions their
leaching rate, which is very very low, can increase. If conditions pre-
vail which cause humic acids to migrate, i.e., addition of urea, some
evidence exists which shows humic acid-DDT combinations will leach down
through soil. The effect is minor and the conditions very uncommon.

Several of the classes of compounds listed in the baseline groups can be
expected to show volatility. Especially in spray irrigation, it is safe
to predict that some of the organics will be found far removed from the

- 107 -

spray irrigation area. This does not really cause a problem since the effect is to dilute the amount reaching the soil in a given area, and as the above review shows, very few of the organics are of potential stability when exposed to soil microorganisms.

Other means of control of organics may occur -- several modes of control such as ion exchange and adsorption can be expected to retain several of the classes of compounds. The problem class, polynuclear hydrocarbons, is one of these. These are relatively large molecules and are strongly retained by river muds, etc. They can be expected to be of very limited mobility in a soil water system.

CHAPTER X - IMPACTS OF TREATED WASTE WATER

CHAPTER X - IMPACTS OF TREATED WASTE WATER

Application of waste water at a rate of two inches per week would require
12.8 acres per 1000 population thus, a city of 1/2 million
population would require 10 square miles of waste water disposal site,
assuming a per capita production of 100 gallons per day. Land areas
of this magnitude are not available adjacent to major population centers.
This suggests a location in rural agricultural areas or semi-wild or
forest land. Applications of this quantity of water would have signifi-
cant influences on several phases of the natural ecosystems. The quan-
tity of water applied (100 inches/year) to the disposal site would have
the influence of an equivalent amount of rain spaced at weekly intervals.
The effects of 100 inches of precipitation on climates which currently
receive 40 inches or less will drastically alter processes of soil forma-
tion, thus soil properties. Thermal regimes of the particular micro-
climates will also be altered. Factors which will significantly alter
climate will also alter vegetation and hydrologic regimes.

A. Environmental Effects on Ecosystems

The following sections consider certain gross effects of waste water
disposal at the design loadings on factors of the environment and their
interaction in ecosystems.

Climate: Both the spray irrigation method and overland runoff would have
major effects on microclimate when applied to large contiguous areas (2-10
square miles). Both of these methods would apply sufficient water to a
site so that annual evapotranspiration losses would proceed at the poten-
tial rate. Evapotranspiration of water utilizes energy through latent heat
of vaporization. Incoming solar radiation in the California study area has
sufficient potential energy to evaporate 60-80 inches of water annually.
Assuming a maximum incoming radiation of 600 to 800 langleys per day and
near perfect absorption of this incoming radiation by the vegetative crop,
evapotranspiration would utilize about 0.4 of an inch of water per day. It
has been observed in irrigated areas that this evaporation rate is sufficient
to reduce maximum air temperatures 15-20°F.

In other areas or during seasons when evaporation rates are low increasing the moisture content of the soil would increase soil specific heat, thus, yielding colder soils and water to the water table and eventually the surface stream system.

These same influences apply to a lesser degree in both the Lake States and New England study areas as potential evapotranspiration is less. Philipps (1971) suggests that spray irrigation adjacent to "one of the better trout streams in the country" has resulted in no measurable change in water quality. He has not reported, however, if his concept of water quality includes thermal regime.

In general the application of water by either overland runoff or spray irrigation should also moderate cold winter climates. Water has a very high specific heat, thus, will absorb large quantities of incoming radiation with less change in temperature. In a like manner the stored heat will be released during cool periods to maintain higher minimum temperatures. While increasing the quantity of water available for evaporation might have an overall impact of reducing soil temperature, increasing soil water content generally has the overall impact of increasing the conductivity of heat (Baver, 1948). The overall impacts of the design waste water loading on large areas of watershed should be one of reduced temperature gradient between the evaporating soil surface and percolating water at depths of 3-5 feet. In addition the potential for transfer of sensible heat from the waste water disposal area would be reduced. These phenomena are observed in spray irrigated croplands.

Assumptions of effects of evapotranspiration on microclimate for spray irrigation and overland runoff do not apply to rapid infiltration ponds. The land area required for disposal of the same volume of water by rapid infiltration is less than 1/30 that required for spray irrigation or overland runoff (200 acres compared to 6,400 acres). The microenvironment adjacent to the free water surfaces will be altered substantially by growth of riparian vegetation and evaporation from the free water surface. Equal quantities of water will be disposed of in each of the three methods, thus, a shallow soil depth to the water table might stimulate phreatophytes

- 110 -

through an extensive area. Should this happen evapotranspiration by
these plants would have many of the same influences on microclimate as
spray irrigation. The potential for high volumes of rainfall which
would flush waste water through the aquifer system should be considered.
The design rate of flow for rapid infiltration requires a soil-waste
contact time to achieve renovation. A high volume of rainfall might
accelerate flow rates so that the desired degree of renovation would
not be achieved.

An unexpected rain storm of high intensity would also be a definite
potential hazard with overland runoff disposal. Several inches of
rain in a brief period on impermeable soils could flush high quantities
of waste water into the surface collection system. Under these condi-
tions renovation would not be complete and provisions should be made for
storage of this water with later return to the disposal site.

The design of the spray irrigation system is such that the resident time of
the waste water is considerably longer in the soil profile and infiltration
rates would be maintained so that vegetation would be washed by rain of
the waste water effluent, permitting material to infiltrate the soil. If
rainfall rates were sustained and infiltration capacity exceeded, the run-
off water would be relatively pure rainfall.

Soils: Changes in microclimate along with the addition of effectively
100 inches of precipitation will significantly alter soil formation pro-
cesses particularly in California under spray irrigation. Soil pH and
base exchange chemistry would be particularly influenced. Rapid infil-
tration ponds and overland flow would not influence surface soils. (See
Chapter IX)

Soil chemical properties are definitely influenced by the balance between
total precipitation and total evapotranspiration. When the quantity of pre-
cipitation exceeds the evaporation water soluble chemicals are leached
from the soil profile. When the quantity of precipitation is less than
evaporation soluble salts accumulate at varying depths in the soil profile.
Both the New England and Lake States study areas have precipitation which
sufficiently flush alkaline, water soluble salts from the soil profile.

Many soils in the California study area accumulate certain water soluble
salts as annual precipitation is not sufficient to wash these salts from
the soil profile. Application rates of two inches of water per week
would not be sufficient to leach soluble salts from the soil profile
during the summer months in California. Winter rains combined with
application rates of 2 inches per week would be sufficient to pass water
through the soil profile starting in October. If waste water loadings
are restricted to 2 inches per week then large quantities of soluble
salts will be leached to the water table and streams each fall. Coastal
regions of California as represented by Ferdale would provide less
problem with accumulation of salts in the profile during the summer as
the evaporation rate is less than 1/2 that of the warmer arid region
near Davis and Sacramento.

Vegetation: Vegetation in a natural ecosystem is the end product of inter-
actions of climate and soils. If natural vegetation was allowed to grow
in the disposal sites a plant succession would be initiated which would
greatly favor water loving species. Changes in vegetation have been
documented in several studies (Elazar, et al, 1971). The most dramatic
influences on vegetation could be expected in California where the most
drastic alteration of climate and soils would take place.

Hydrologic Regimes: The hydrologic regime of a given area is the effects
of climate acting on vegetation, soil and geologic parent materials. The
gross influences of waste water disposal by both spray irrigation and over-
land runoff drastically alters climate, vegetation and thus, will have very
significant impacts on hydrologic regimes. Monthly precipitation may be
visualized as increased by the amount of applied waste water. Flow regimes
graphically illustrated in Figures III-1, 2, and 3 could have monthly
rainfall increases of approximately 8 inches, thus removing any moisture
deficit from the Lake States or New England regions and contributing large
additional amounts of water to surface streamflow.

Eight inches of water per month would be contributed directly to groundwater
and streams during seasons of low evapotranspiration loss in both the Lake

States and New England. Winter streamflow rates would be quadrupled in small streams of these areas. Increases in summer flow rates would also be significant though not predictable as the balance between evapotranspiration and streamflow cannot be assessed accurately. Streams in the proposed disposal area in California have very low flows now and frequently are dry through much of the summer. The proposed waste water application rates would quadruple winter streamflow and probably be sufficient to maintain flow during the dry summer months.

The potential exists for the interaction of leachate chemistry with suspended solids in streams. A study of stream recovery (EPA, 1971) after manganese strip land reclamation reveals that some of the primary effects of reclamation are related to turbidity rather than toxic effects of high levels of manganese. Dissolved chemistry has the ability to associate with suspended sediment or organic residues thus, altering the chemistry of the suspended material. Manganese, iron and aluminum are lost principally in association with suspended soil particles. Frederickson (1971) reported the loss of organic nitrogen on suspended sediment averaged 54% of the total loss following forest clearcurring, (3.4 lbs/acre/year).

In all areas a potential for increased flooding exists as a result of waste water loading. The regulation of flood runoff from small drainages is an important problem in many areas around the San Francisco region. If substantial portions of relatively small watersheds receive waste water loadings in addition to natural winter storm rainfall substantial increases in flood peaks might occur. This phase should be studied by simulation of hydrologic regimes. Exact evaluation of flood potential is dependent on several factors of which the degree of soil cover is of prime importance. Native vegetation is frequently sparse in the arid foothills of California. High intensity rainstorms of long duration destroy surface soil aggregates, thus plugging soil pores and reducing infiltration capacity. Under these conditions high volumes of surface runoff occur quite rapidly contributing to erosion and flood peaks. A spray irrigation system which maintained a complete vegetative cover as

well as high infiltration rates might reduce flooding potential. Over-
land runoff disposal systems on relatively impermeable soils would con-
tribute substantially to the flood potential.

B. Chemical Effects on Ecosystems

Nitrate in Domestic Water: The USPHS in 1962 recommended a limit of 45
mg/l nitrate as NO_3 (10.17 mg/l NO_3-N) in drinking water. The recommended
limit for European drinking water was set in 1961 by the World Health
Organization at 50 mg/l NO_3 (11.3 mg/l NO_3-N). These limits were re-
commended because of the relationship between high nitrates in drinking
water and nitrate poisoning in infants (infant methemoglobinemia) (McKee
and Wolf, 1971). Low removal efficiencies of nitrates from waste water
in a land disposal system could result in contamination of the groundwater
and present a potential health hazard if the water is reclaimed for un-
restricted municipal use. Although the concentration (if all converted
to NO_3-N) in the model waste water (20 mg/l) is almost twice that of the
recommended limit for drinking water, maintenance of removal efficiencies
commensurate with well managed land disposal systems (efficiencies of
nitrogen removal $\geq$ 80%)   coupled with dilution from rainfall (See Table
VII-1) and groundwater, however, would insure against a health hazard.
The highest concentration reaching groundwater, as much as 6 mg/l, would
occur in the California area as a result of the concentrating effect of
evaporation. In poorly managed situations removal efficiencies could be
as low as 50% which would result in NO_3-N content exceeding the 10 mg/l
limit.

Eutrophication: A potential impact of land disposal of secondary waste
water on the surrounding environment is accelerated eutrophication of
contiguous surface waters. Accelerated eutrophication can occur in either
standing or running water, but is most well known in standing water.
However, in both environments planktonic or attached plants assimilate
available nutrients according to their requirements.   These requirements
are in part dictated by their finite capacity for uptake and other grow-
ing conditions such as light and temperature. If growth and population

increase in either standing or running water is limited (or restricted)
by the quantity of an available nutrient, then addition of such a
nutrient will stimulate growth and result in a larger mass of algae,
possibly reaching nuisance proportions if the nutrient increase is
great enough. Large masses of nuisance plants indicate accelerated
eutrophication.

The two nutrients most often cited as causes of accelerated eutrophi-
cation are N and/or P. Planktonic algal growth can be limited by many
nutrients, including carbon, iron, and several other trace elements.
However, a principal reason why N and P, and particularly P, have been
most often linked to accelerated eutrophication is that, relative to
needs of algae, these elements are the most scarce naturally. Phosphorus
probably has more "sinks" and fewer sources than nitrogen, so is most
often considered the nutrient most responsible for long term increase
in algal mass and productivity which is characteristic of eutrophica-
tion.

Significantly increased loading of N and/or P would be most apt to result
in nuisance levels of algae if added to standing water than if added to
running water. This is probably more true for shallow than deep lakes.
Even though growth is possible in the lighted surface water only, the
incoming nutrients are usually efficiently scavenged and converted to
algal mass. Because current is very effective in maintaining transport
of nutrients to the bottom attached algae in streams growth is often
maximized at lower concentrations than in lakes. Thus, further increase
in nutrients in streams has little effect except when existing concen-
trations are very low or current is very sluggish. However, if travel
time is sufficiently long for nutrient content to be eventually depleted
by uptake, then plant growth could again be limited downstream. But in
the meantime nutrients have been distributed over a long distance, parti-
cularly if flow is great. In the case of sluggish flow or standing water,
nutrients are detained allowing ample time for more complete utilization
in a smaller area, thus, larger algal masses are apt to result from in-
creased nutrient concentrations.

The impact of treated waste water on eutrophication of contiguous surface
waters will be considered from the standpoint of total annual loading and
possible concentration increase of N and P. Vollenweider (1968) has re-
viewed trophic characteristics of several lakes in the world and,
according to their measured N and P loading rates and mean depths, has
suggested admissable and danger limits for these two nutrients. By
significantly exceeding the danger limits in annual loading of N and/or
P in a lake of known depth, eutrophication will probably be accelerated.
However, the rate of acceleration cannot be estimated from his relation-
ships. Although predictive techniques for the eutrophication process
are in their infancy, these relationships can serve as management guide-
lines.

An area of 10 square miles with 2 inches of model waste water applied per
week was used as a guide to estimate nutrient loading to contiguous water
bodies from groundwater recharge or overland runoff. This represents a
facility that could serve 5×10^5 people and conforms to that used to
estimate physical effects on ecosystems. Table X-1 shows the mean con-
centrations of N and P in waste water and rainfall combined along with the
total water applied to the area by both processes. P in rain was neglected.
Evaporation was not subtracted from total quantity of water added. Apply-
ing the factor in Table X-1 and the lowest estimated treatment efficiencies
provides estimates of the greatest probable N and P loading reaching surface
waters via reclaimed water from the three application techniques and areas
in the country (Table X-2). Note that P loading will probably be greatest
using overland runoff and least using spray irrigation. From Table X-2
it is also clear that loading to standing water from land disposal may
produce a significant effect with N because in two techniques of disposal
the ratio of N/P by weight is less than that required by algae, 7.2/1.
Relatively, more P would be available than N. With spray irrigation the
opposite is true which is due to the high efficiency of P removal by that
technique.

Considering Vollenweider's (1968) suggested danger limits and standing
water with a mean depth of 65 feet, Table X-3 gives estimates of the area

- 116 -

Table X-1

ESTIMATE OF CONCENTRATION OF N AND P APPLIED TO 10 MILE^2
LAND DISPOSAL SITES (SERVING ONE HALF MILLION PEOPLE) IN THREE
AREAS AND WATER VOLUME THAT REACHES GROUNDWATER CONSIDERING MODEL
WASTE WATER AND APPLICATION RATE OF 2 INCHES PER WEEK

Area and application period (mos)	Fraction of water added from two sources		Med. conc. in rain in mg/l	Mean conc. in mg/l in rain and waste-pre-treatment		Infiltrate to ground water in inches	Factor for conversion to lbs./ac/yr X 10^5
	rain	waste	N	N	P		
Great Lakes (9)	.32	.68	.59	13.8	10	115	1.82
New England (8)	.50	.50	.35	10.2	10	140	2.04
California (12)	.13	.87	.38	17.5	10	120	1.75

Table X-2

ESTIMATED LOSS OF N AND P TO GROUNDWATER FROM 10 MILE^2 AREAS IN
LBS./ACRE/YR. X 10^5 (FACTOR X CONC. IN TABLE X-1) USING LOWEST
PROBABLE TREATMENT EFFICIENCY (TABLE XI-1) ASSUMING WELL
MANAGED APPLICATION OF MODEL WASTE WATER AT 2 INCHES PER WEEK

Area and Application Period (mos)	Spray Irrigation		Overland Runoff		Rapid Infiltration Ponds	
	N (.80)	P (.99)	N (.80)	P (.70)	N (.80)	P (.90)
Great Lakes (9)	5.0	.18	5.0	5.5	5.0	1.8
New England (8)	4.2	.20	4.2	6.1	4.2	2.0
California (12)	6.1	.18	6.1	5.3	6.1	1.8

Table X-3

ESTIMATED 1000'S OF ACRES OF STANDING SURFACE WATER AVERAGING 65 FEET
IN DEPTH THAT MAY BE NECESSARY TO DISSIPATE THE ANNUAL N (37 lbs/
ac/yr) AND P (2.7 lbs/ac/yr) LOAD THAT WOULD REACH THE SURFACE
THROUGH GROUNDWATER RECHARGE WITHOUT CAUSING UNNATURALLY
ACCELERATED EUTROPHICATION. ESTIMATES BASED ON TENTA-
TIVE DANGER LOADING RATES FOR EUTROPHICATION PRO-
POSED BY VOLLENWEIDER (1968)

Area	Spray Irrigation		Overland Runoff		Rapid Infiltration Ponds	
	N	P	N	P	N	P
Great Lakes	13.5	6.7	13.5	204	13.5	67
New England	11.3	7.4	11.3	226	11.3	74
California	16.5	6.7	16.5	196	16.5	67

in 1000's of acres that would be required to dissipate the nutrients
short of causing accelerated eutrophication. The danger limits in
lakes are 37 lbs. N/acre/year and 2.7 lbs. P/acre/year. The lower
estimate of treatment efficiency by overland runoff results in the largest
estimate of area needed for nutrient dissipation -- about 200,000 acres
if the groundwater drainage from such a 10 square mile disposal site
reaches standing surface water average 65 feet deep. Even with spray
irrigation, nearly 7,000 acres are required. Of course loss of nutri-
ents from the site to surface water through groundwater or seepage from
overland runoff is not the only path. If erosion from the site sur-
face occurs, which is typical in agricultural areas, this could be
another important path for nutrient loss to surface water. Even though
treatment efficiencies for N and P removal are relatively high by most
techniques, sizable loads may still escape and if the ground or surface
water runoff reaches nearby lakes, accelerated eutrophication may occur
if the lakes are not large or deep enough for adequate dissipation of
added nutrients.

If concentrations of N and P in groundwater are considered the eutrophi-
cation impact in running water can be discussed. Table X-4 shows ground-
water concentrations of N and P following treatment in the three areas.
Evaporation has been considered in these estimates as is suggested by the
higher concentrations in the more arid California area. If these esti-
mated concentrations in groundwater from disposal sites are compared
with data from streams in Tables IV-2 and IV-3 (convert to NO_3-N and
PO_4-P), it can be seen that for almost all streams the entrance of such
groundwater or overland seepage would increase the concentration of N
in the receiving stream. This is even true for the maximum values in
most cases. The exceptions to this are for maximum NO_3-N in the Maumee
and Sandusky Rivers in Ohio. There, ground or seepage water addition
would significantly dilute existing concentrations in those rivers.
Maximum P concentration would be reduced in all streams where P data are
given. This is particularly true for reclaimed water from spray irri-
gation. It should be noted that at those concentrations in rivers (up to

- 120 -

Table X-4

ESTIMATED CONCENTRATIONS OF N AND P IN MG/L IN RENOVATED WASTE WATER

Area	Spray Irrigation		Overland Runoff		Rapid Infiltration Ponds	
	N	P	N	P	N	P
Great Lakes	3.5	0.12	3.5	3.8	3.5	1.2
New England	2.7	0.13	2.7	3.9	2.7	1.3
California	6.4	0.19	6.4	5.6	6.4	1.9

8 mg/l N and 0.6 mg/l P) algal growth is no doubt saturated for the
light conditions present. Even with dilution of nutrient concentrations
in those two streams with groundwater containing 3.5 mg/l N and 0.12
mg/l P (spray irrigation) algal growth or mass will probably not be
affected. Phosphorus data are incomplete from streams in the areas, but
in several instances NO_3-N is apparently below or approaches 0.1 mg/l
for much of the time. This is true in the Merrimack, Miller, Putah,
Bear and Feather Rivers. In those streams it appears likely that ground-
water containing 3.5 mg/l N could stimulate growth if light and temper-
ature are optimum.

Toxicity - Water: The concentration of toxic heavy metals in reclaimed
waste water would not be expected to be sufficiently high to adversely
affect organisms in receiving streams. Concentrations in waste waters
are all 0.2 mg/l or less and even in seepage waters from overland runoff,
where relatively low removals are expected, no effect would be expected.

Toxicity - Soils: The potential for toxic levels of heavy metals to occur
with sustained applications of waste water might be inferred from the
management of agricultural soils where heavy metals have been added as
pesticides and substantial quantities of irrigation water applied. Past
orchard management practices in the Wenatchee, Washington area have
added significant amounts of zinc, copper and lead arsenate to soils.
Average active application rates in lbs/acre/year are: Zinc, 30, copper,
10, lead, 160, arsenic, 50. Use of these heavy metals has largely been
discontinued, however, they were applied consecutively to the same acres
for over twenty years. Usual irrigation practices applied 3-5 feet of
water per year.

The total amounts of lead applied would exceed 3,000 lbs. The application
rate suggested in the model effluent would apply a maximum of 0.7 of a
pound per acre per year. Thus, it would take almost 5,000 years to apply
an equivalent amount of lead through waste water. Similar calculations
can be made for other heavy metals which demonstrates that the application
rates based on our knowledge of soil chemistry and equilibrium conditions.

Fine textured soils (silt loam) should have a capacity to absorb heavy
metals for an indefinite number of years. Water leaching from these
soils should have a quality at least equal to the composition of surface
water currently flowing in the study areas.

CHAPTER XI - SUMMARY AND RECOMMENDATIONS

CHAPTER XI - SUMMARY AND RECOMMENDATIONS

The effectiveness and expected mechanisms of renovation of waste water
disposal methodologies are summarized in Tables XI-1, 2, 3 and 4.

A. Effectiveness of Land Disposal Techniques

Removal of BOD and SS in spray irrigation and rapid infiltration is
expected to be very effective -- 99 percent. This is largely due to
the high filtering effectiveness of the soil mantle and maintenance
of aerobic conditions by alternating wet and dry periods. These re-
moval efficiencies have been attained with loading rates thousands of
times greater than proposed here, so nearly complete removal is expected.

Overland runoff is expected to provide a higher residual in the seepage
water leaving the field because some of the filtering capacity of the
soil is lost with horizontal transport of the waste water. Since sus-
pended solids are largely organic in secondary waste water, their be-
havior should about parallel that of BOD.

Efficiencies of N and P removals are based on values obtained the last
15-20 years in comparable experimental systems under a variety of en-
vironmental conditions. The percent N removal (80-90%) is in good agree-
ment with agricultural fertilization experiences. The low efficiency
of P removal in the overland runoff system is attributed to short reten-
tion time of waste water on soils. The indicated higher P percent leached
to groundwater in the rapid infiltration ponds is based on the lower
fixation capacity of coarse textured soils.

Efficiencies of N removal may be increased (greater than 90%) and maintained
for many years by additions of organic carbon. A high carbon-nitrogen ratio
will accumulate nitrogen in the soil reducing leaching. Increased soil
organic matter will also maintain or improve soil structure, increase the
potential for plant uptake and aid aeration and water flow processes.

Heavy metals are efficiently renovated by spray irrigation as 100% of the
waste water will infiltrate the soil column, thus allowing both time and
surface area for chemical reaction.

Table XI-1

ESTIMATED EFFECTIVENESS OF LAND DISPOSAL TECHNIQUES

	Spray Irrig.	Overland Runoff*	Rapid Infil. Ponds
BOD	99	80	99
SS	99+	80	99
N	80-90	80	80
P	99	80	90
Heavy Metals	99	10-30	95
Organic Comp.	99	50	90
Viruses	99+	90	99+
Bacteria	99	90	99+
Total Dissolved Solids			
Cations	0-75	0-50	0-75
Anions	0-50	0-10	0-50

*Values are best estimates in light of limited information

Spray Irrigation Condition

Soil: silt loam
5' deep
$C/N \geq 15$

Climate: Great Lakes
2" in 8 hrs. once a week

Continuous Vegetation

Overland Runoff

Application: Intermittent

Soil: low infiltration
clay or clay loam
less than .2"/day

Effective Travel Distance: 150'

Slope: 2 to 6%

Continuous Vegetation

Rapid Infiltration Pond

Application: 330'/yr.
flood up to 10 days, 4 days dry
alternate aerobic and anaerobic

Soil: sandy gravel

Effective Travel Distance: 200'

Continuous Vegetation

Table XI-2

RELATIVE IMPORTANCE OF RENOVATION MECHANISMS - SPRAY IRRIGATION

	a. Plant Up-Take	b. Fixation	c. Volatilization	d. Ion Exchange	e. Filtration
BOD	0	2	3	0	4
SS	0	2	2	0	4
N	4	1	4	1	0
P	4	3	0	2	0
Heavy Metals	1	4	0	4	0
Organics	0	0*	4	0	1
Viruses	0	2	3	0	4
Bacteria	0	2	3	0	4
Total Dissolved Solids					
Cations	0	4	0	2	0
Anions	2	4	0	1	0

*Fixation is significant for chlorinated and polynuclear hydrocarbons

Scale: 0 to 4
        "0" relatively insignificant
        "4" major mechanism

a. removal by harvested crops

b. adsorption, precipitation, absorption

c. loss to atmosphere via formation of CO_2, N_2, NH_3, NO_4

d. as normally defined, assumed reversible

e. physical retention in soil

Table XI-3

RELATIVE IMPORTANCE OF RENOVATION MECHANISMS - OVERLAND RUNOFF

	a. Plant Up-Take	b. Fixation	c. Volatilization	d. Ion Exchange	e. Filtration
BOD	0	2	4	0	3
SS	0	2	2	0	4
N	2	1	4	1	0
P	2	4	0	1	0
Heavy Metals	0	4	0	2	0
Organics	0	0	4	0	1
Viruses	0	2	4	0	3
Bacteria	0	2	4	0	3
Total Dissolved Solids					
Cations	0	4	0	0	0
Anions	0	4	0	0	0

Scale: 0 to 4
         "0" relatively insignificant
         "4" major mechanism

a.   removal by harvested crops

b.   adsorption, precipitation, absorption

c.   loss to atmosphere via formation of CO_2, N_2, NH_3, NO_4

d.   as normally defined, assumed reversible

e.   physical retention in soil

Table XI-4

RELATIVE IMPORTANCE OF RENOVATION MECHANISMS - RAPID INFILTRATION PONDS

	a. Plant Up-Take	b. Fixation	c. Volatilization	d. Ion Exchange	e. Filtration
BOD	0	2	2	0	4
SS	0	2	2	0	4
N	0	0	4	0	0
P	0	4	0	0	0
Heavy Metals	0	4	0	2	0
Organics	0	0	4	0	1
Viruses	0	2	3	0	4
Bacteria	0	2	3	0	4
Total Dissolved Solids					
Cation	0	4	0	0	0
Anion	0	4	0	0	0

Scale: 0 to 4
       "0" relatively insignificant
       "4" major mechanism

a. removal by harvest crops

b. adsorption, precipitation, absorption

c. loss to atmosphere via formation of CO_2, N_2, NH_3, NO_4

d. as normally defined, assumed reversible

e. physical retention in soil

Overland runoff does not permit sufficient contact for heavy metals to exchange or react chemically in the soil to obtain the degree of renovation expected with rapid infiltration or spray irrigation.

Trace metals will be renovated approximately 95% by rapid infiltration ponds assuming soils have a few colloids and the system is reasonably aerobic.

In general the several classes of organic compounds reported in waste water are subject to microbial decomposition with very few exceptions. These exceptions are humic substances and lignins, which are only slowly degraded, chlorinated hydrocarbons which are essentially stable to degradation, and polynuclear hydrocarbons which appear to be stable. All of these latter classes of compounds appear to be held in soil. (i.e., They do not seem to penetrate into the water table in any appreciable amount, they all are subject to runoff into streams.)

Spray irrigation is clearly the best method of application as far as organics are concerned. Considering the baseline criteria for concentrations of the various classes of organics, and the criteria for spray irrigation application, aerobic conditions will be maintained enough of the time, and the essential penetration of the organics into the soil will be accomplished. This gives adequate contact and retention time with soil microorganisms for decomposition to occur. There are not critical differences in the several study areas peculiar to organics applied by spray irrigation.

Overland runoff is the least desirable method of application, especially if one is dealing with an effluent with over-baseline values of chlorinated hydrocarbons, oils, polynuclear hydrocarbons and perhaps humic substances. The runoff method decreases chances for penetration of these organics into soil and may result in anaerobic conditions more of the time. Only limited information on this method of application is available and most of that is in very warm climates. This method probably would be less effective in cooler regions. An estimated degree of effectiveness of removal of or-

ganics of only 50% is given for this application method, partially for
the above reasons and partially because the study calls for secondary
treated effluent, in which the remaining organic compounds are the more
resistant to degradation, and require contact time with the soil.

The rapid infiltration method of application is intermediate in effective-
ness in the three categories reviewed. The method seems to be nearly
as effective as spray irrigation for organic removal.

Bacteria and viruses are removed from waste water systems by the upper
soil mantle (1 cm - 7 feet depending on soil) very efficiently when applied
by the spray irrigation and the rapid infiltration pond systems of
disposal. However, when processed by the overland runoff method, viruses
and bacteria will probably be removed somewhat less efficiently (90% vs
99%).

Although the waste water in an overland runoff system flows a few hundred
feet through and over the top few inches of soil and through the litter
and vegetative cover, there is a greater possibility for the bacteria and
viruses to be carried to receiving waters by the overland flow method
than by methods utilizing vertical percolation of the waste water through
the soil.

Total dissolved solids as ions will be most effectively renovated by
spray irrigation or rapid infiltration ponds. The residence time in
association with an adequate surface of colloidal soil particles should
be sufficient to achieve the degree renovation suggested. Overland runoff
will be less efficient in removing total dissolved solids as only the
soil surface and impeding plant residues are effective in the chemical
reaction.

B. Importance of Renovation Mechanisms

Filtration is the most important removal mechanism for BOD and SS, parti-
cularly where the waste water is allowed to percolate vertically through
the top several inches of soil mantle. This removal is largely physical,

- 130 -

especially where non-degradable SS are concerned. However, dissolved BOD and the filtered particulate organic matter will be decomposed by aerobic microorganisms and largely lost to the atmosphere (volatilization) as CO_2. This process must account for most of the loss because with spray irrigation and BOD loading thousands of times greater than proposed here, little sludge accumulation occurs in the profile if aerobic conditions are maintained.

Volatilization may be, therefore, relatively less important in rapid infiltration because anaerobic conditions may be more likely.

Filtration is relatively less important than volatilization with overland runoff because of horizontal transport of the waste water and a larger BOD residual is expected with overland runoff. Volatilization may be slightly less important at removing suspended solids than BOD because of the non-decomposable component of SS.

Plant uptake of N and P and the subsequent removal of the harvested crop can be a major mechanism of N and P removals in both spray irrigation and overland runoff systems. With the latter system the plant uptake mechanism is perhaps less significant while in the case of rapid infiltration ponds, where the plant material is not normally removed by harvesting, this mechanism does not contribute to N removal since the plant N is returned to the system from the decomposition of plant remains.

Fixation of N as NH_4^+-N or, to a lesser degree, as organic nitrogenous compounds in clay minerals or through immobilization of N in humic matter or microbial cells has not been reported as a significant mechanism of N removal in either the spray irrigation or overland runoff systems and for all practical purposes it is completely absent in the rapid infiltration ponds. The P removal by fixation is one of the major mechanisms in all three waste water application systems. In rapid infiltration ponds where sand may not contain enough P reactive surfaces, incomplete removal of P may be anticipated. Volatilization of N gases such as N_2 or N-oxides

through biological nitrification-denitrification remains the major mechanism of N removal under all three waste water application systems.

Removal of N and P by ion exchange is of relatively little importance in the spray irrigation and overland runoff systems and of no value in rapid infiltration ponds.

Heavy metals will be largely removed from the waste water using spray irrigation through ion exchange and fixation in the soil. In the overland runoff disposal method heavy metals will be fixed in association with organic matter and on surface soil particles as water trickles over the impermeable soil. Heavy metals also will be fixed on soil particles in rapid infiltration disposal and to a lesser degree via ion exchange.

Renovation of total dissolved solids as represented by cations and anions will be accomplished primarily by fixation with relatively insignificant amounts of plant uptake and possibly slightly more ion exchange. Ion exchange, however, should not be considered a renovation, but rather a substitution of one ion complex for another as the systems must maintain electrical neutrality.

It appears that mechanisms of virus and bacteria removal are essentially similar in soil systems treated with waste water. However, there is a difference in the relative ranking of the various mechanisms when the methods of disposal are considered. Volatilization in overland runoff systems appears to be most effective because the vertical filtration effectiveness of soil is not utilized. Organisms that are removed will be largely a result of decomposition and CO_2 evolution. Filtration is most important in rapid infiltration and spray irrigation because the vertical filtering capacity of the soil can be used. Fixation, or the accumulation of organic matter in cells or adsorbed onto inanimate particles, is of less importance.

All factors considered, spray irrigation disposal of waste water is re-
commended. Application rates of two inches per week should maintain
aerobic soil conditions and expected renovation efficiencies in many
soils common in each study area. A vigorous growing of continuous cover
crop is necessary to maintain soil infiltration rates, prevent soil ero-
sion and provide organic matter to the soil. If the crop is harvested,
nitrogen renovation should be improved.

Overland runoff appears to be a more difficult disposal system in these
five study areas. Extensive soils of a nearly impervious nature do not
occur. Renovation of waste water is accomplished largely on the soil
surface suggesting both a limited capacity (or short number of years)
for renovation and a potential hazard from high intensity rainstorms
flushing the disposal area.

Rapid infiltration ponds would probably be the most difficult of the
three disposal systems to manage and monitor. Anaerobic conditions im-
prove nitrogen renovation, but also develop a complex of other soil,
chemical and physical problems. Reducing chemical conditions change
many processes and provide for slime growth which clogs soil pores.
Often the movement of subsurface water is not known, making monitoring
of the system difficult. Chemical renovation by soil is less pre-
dictable due to the coarse textures required for rapid soil water movement.

C. Recommendations for Future Study

In the professional judgement of the principal project participants
several aspects of soil-waste water interactions occurring in land
disposal methodology are in need of more extensive research. These
are:

1. The degree of removal of microorganisms, BOD and suspended solids
    as a result of overland runoff procedures should be more thoroughly
    evaluated.

2. The effectiveness of wind transport of pathogens should be examined
    for typical spray irrigation, overland runoff and rapid infiltration
    practice.

3. The potential for nitrogen and phosphorous transport to surface waters
    by the erosion process should be evaluated, particularly for the
    overland runoff procedure.

4. Minimum distances should be established for the separation of dis-
    posal sites from receiving surface waters or other points of water
    use.

5. Simulation models should be developed for the purpose of evaluating
    ecosystem response particularly for the hydrologic regime.

6. Soil-chemical interactions of soil forming minerals and heavy metals
    in various major soil types should be investigated. This should
    include the relationships of waste water-soil interactions and the
    potential development of secondary minerals.

7. The potential exists for improved nitrogen renovation through manipu-
    lation of the C/N ratio on spray irrigation. Interaction of soils-
    climates and loading rates should be investigated.

- 134 -

8. More extensive research is needed concerning the optimal loadings
    of nitrogen and phosphorous in each land disposal method as a function
    of climatic and vegetative conditions.

9. Future research should not be confined to design criteria assumed
    in this study. The type of effluent (secondary or tertiary treated)
    loading rate and type of ecosystem (forest, agricultural, pastureland)
    should be treated as variable for future study emphasis.

10. The costs (financial and social) associated with all forms of land
    disposal techniques are in need of much more extensive evaluation.

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-147-

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